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Vegetation Changes in Riparian Zones Following Wildfire in Southwestern Alberta

by

Nicole L. Salamon

A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of Master of Science

in

Water and Land Resources

Department of Renewable Resources

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Date: Sept. 11, 2003

Abstract

In the winter of 1997, a wildfire burned across native grassland landscape in southwestern Alberta. Riparian zones in this area are very important for livestock and wildlife forage, water and shelter. From field observations significant differences in percent woody vegetation cover between paired burned and unburned riparian zones were detected in 2002. Pre- and post-fire Landsat data were also acquired to monitor vegetation cover differences between paired burned and unburned riparian zones. Techniques to characterize spectral data, isoclustering, principle component analysis and normalized difference vegetation index, and patch statistics were applied to analyze changes in vegetation differences over time. The spectral differences between the burned and unburned areas increased immediately following the fire then decreased over time. The ability to identify specific vegetation differences over time was limited by a combination of the spatial resolution of Landsat image data, the variable grazing management strategies, and weather patterns.

We are uniquely fire creatures on a uniquely fire planet. To study fire is to inquire into one of the informing processes of the earth; to manage fire is to perform one of the defining acts of human beings. ~Steve Pyne~

Dedication

This thesis is dedicated to my parents Lina and Norbert Salamon, my sisters Alexandra Salamon and Elizabeth Morrison, and my niece Mikhayla Morrison, whose continual encouragement and love motivated my passion for nature and learning while I paddled my own canoe.

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Chapter 1 Introduction

1.1 Background

On December 14, 1997 a wildfire referred to as the Granum Wildfire burned across 18,000 ha of native foothills rough fescue grassland and 4,000 ha of cultivated land in southwestern Alberta (Bork *et al.* 2000) (Figure 1.1). The fire occurred mainly on private land west of Fort McLeod off Highway #2 at approximate latitudes 49°40' to 49° 53' North and longitudes 113° 30' to 113° 55' West.

Rangeland is a very important cultural and economic resource on the Albertan landscape (Fitch and Adams 1998, Walker 1993, Hill *et al.* 2000, Huntsinger and Hopkinson 1996). The dynamic interactions between topography, weather patterns, vegetation distribution, animal distribution and disturbance regime in these grasslands are a source of rich biodiversity (Hill *et al.* 2000, Walker 1993). Since settlers first introduces sedentary grazing on the prairies in the late 1800s (Fitch and Adams 1998), ranching has been the predominant land management practice with current livestock ranching operations on native range or cultivation for crops, occupying more than 70% of the Fescue Prairie in Alberta (Hill *et al.* 2000) and much of the western United States (Fleischner 1994). The primary values of rangeland include cattle and wildlife grazing, foraging and browsing, however, provisions for watershed protection and wildlife habitat are also fundamental values (Willms and Jefferson 1993). Preservation of the ecological integrity of rangeland is essential for the economic and cultural sustainability of the people who live there, and also for the flora and fauna cohabiting that landscape.

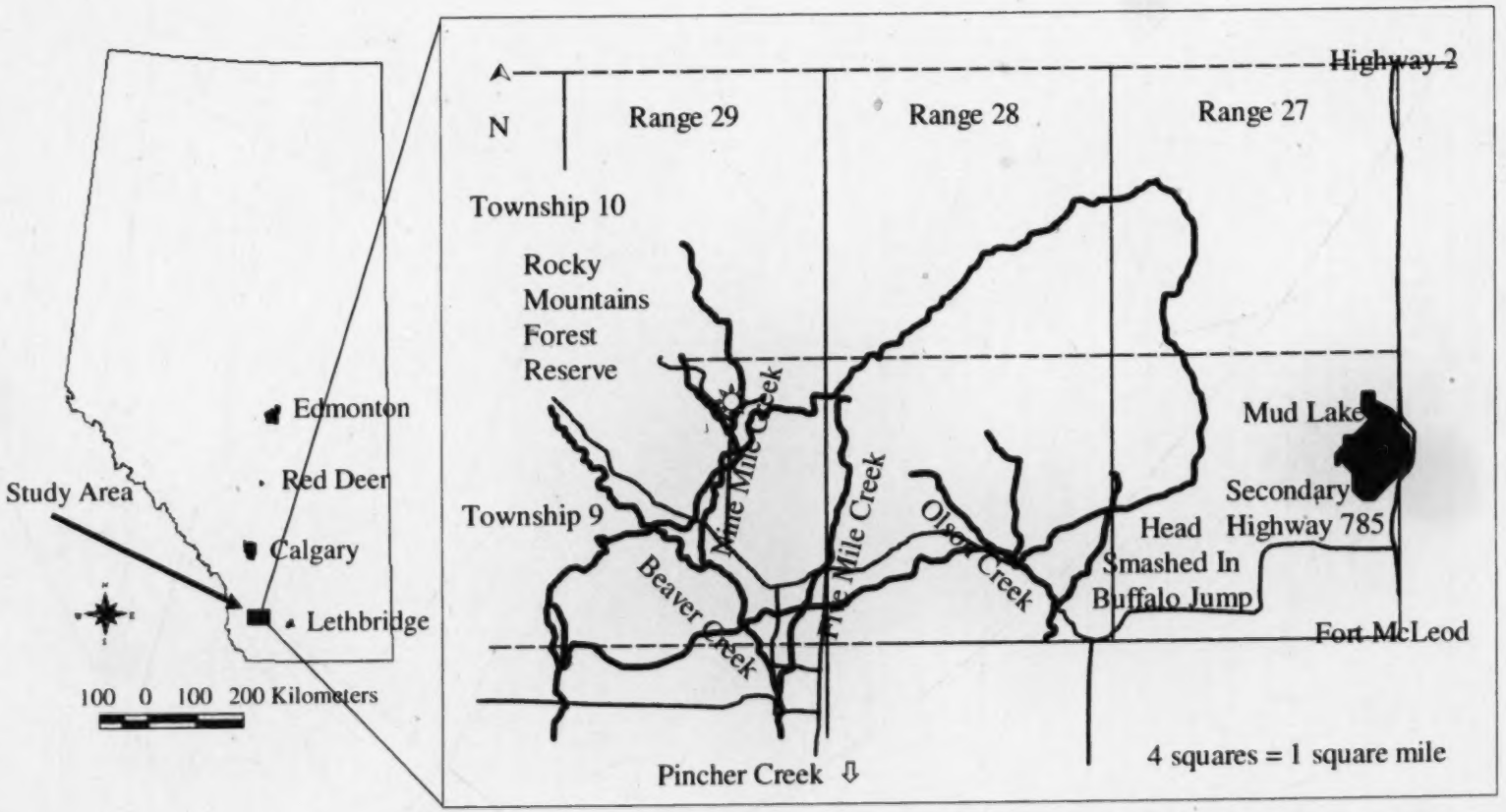

Figure 1.1 Local geographic details of the 1997 Granum Wildfire study area where the red line represents the fire boundary around native rangeland, the blue lines represent major streams and the yellow sun represents the local weather station.

Historically, fire was a natural, frequent and important disturbance in the Fescue Prairie to maintain the ecosystem by removing standing dead vegetation, litter and impeding the spread of woody vegetation. Additionally, fire frees nutrients and space occupied by standing dead vegetation and litter, for use in new vegetative growth (Pyne *et al.* 1996, Vogl 1979). Following fire, vegetation may require a period of relief from grazing pressure to fully regain high productivity, vigour, and community stability. Recovery of both upland range and lowland riparian zones is essential to sustain wildlife and cattle populations in fire-disturbed areas. Forage scarcity immediately following fires temporarily displaces grazers but the resulting high-quality forage available later in the following growing season may attract heavy forage use. This grazing pressure may prevent recovery of plant community composition and productivity to sustainable pre-fire foraging levels. Current land use practices do not accommodate the historical fire disturbance regime because the land has been fragmented by human activity preventing large fires. In addition, the landowners and their animals use the land every year (Willms *et al.* 1985), necessitating continual access to the entire landscape.

Studying post-wildfire landscapes is often difficult because the lack of pre-fire data limits the ability to compare the post-fire vegetation to that on the original landscape (Turner *et al.* 1997). The large extent of the 1997 Granum Wildfire and its remote location created a need to identify an efficient and accurate method to monitor changes in vegetation cover at a landscape-scale. Landsat 7 enhanced thematic mapper (Landsat 7) satellite image data may be appropriate for this type of study. Landsat 7 is beneficial first, due to its 16-day repeat coverage of the landscape allowing for multiple image analyses within the same year or over multiple years. Second, the National Aeronautics and Space

Administration designed the Landsat 7 satellite to collect consistent data and archives are maintained for future reference allowing the user to acquire pre-fire image data (USGS 1999). Thirdly, when studying extensive landscapes, one Landsat 7 image may be sufficient to cover an entire study area whereas other data forms such as aerial photographs or traditional field observations require multiple photographs and labour-intensive data collection, respectively. Also, large area coverage with high spatial resolution satellite data such as IKONOS is more expensive.

1.2 Rationale for Research

Immediately following the 1997 Granum Wildfire a number of research projects were initiated to describe the extent (Spiess and Marciak 1998), the impact (Bork *et al.* 2000) and the severity (Wowk 2000) of the fire across the landscape. In the initial study, the Government of Alberta used Landsat 5 thematic mapper (Landsat 5) satellite image data to complete an initial inventory assessing general land cover prior to the burn and the extent of the burn (Spiess and Marciak 1998). In 1998, Bork *et al.* (2000) used traditional field techniques to evaluate the effect of the fire on the vegetation growth immediately following the fire. In 1999, Wowk (2000) applied Landsat 5 data to evaluate fire severity across the 18,000 hectares of burned native range. Most of these studies concentrated on the more open upland areas of rangeland.

Riparian zones, the subirrigated (overland and groundwater) areas in draws, valleys and surrounding bodies of water, occur within the rangeland. Given that these riparian zones create patches of increased vegetation growth, they are very important features on the landscape. Within these patches, grasses and shrubs are more abundant

and diverse than on the open stretches of adjacent drier rangeland thereby providing more forage, browse, and shelter for animals. Willms (1988) observed greater use of riparian zones by cattle in the spring and early summer than later in the growing season, most likely from greater forage production as a result of more available water. However, Marlow and Pogacnik (1986) observed that when spring precipitation was high, cattle preferred uplands over riparian zones. The greater spring precipitation promotes faster vegetation growth in uplands and because cattle tend to forage where vegetation growth is available, abundant and accessible (Asamoah *et al.* 2003a, in press) and to avoid areas of high soil moisture (Clary and Booth 1993, Mosley 2000), uplands would be preferred. Riparian zones typically either contain water or are closer to the water table (Fitch and Adams 1998) and thus, cattle would need to wade in water or pass through rougher terrain to forage compared to uplands.

Similarly, wildlife such as deer (*Odocoileus hemionus* and *O. virginianus*) and elk (*Cervus elaphus*) use riparian zones for forage, water and shelter (Fitch and Adams 1998). Although riparian zones comprise a very small portion of the rangeland landscape (Asamoah *et al.* 2003b, in press; Fitch and Adams 1998), these areas increase the structural diversity of rangeland with additional edges and vegetation height layers (Fitch and Adams 1998, Gregory *et al.* 1991, Naiman *et al.* 1993). Wildlife use of riparian zones is dynamic throughout the year. During winter, deep snow inhibits efficient use of riparian zones (Peck and Peek 1991). However, in spring, the rapid green up of vegetation provides a rich food source, and during summer the tall, lush vegetation provides shelter for young wildlife from predators. Greater moisture in riparian zones provides increased stability of production supporting active vegetation growth over a

longer period of time than uplands, supplying a food source into late summer and fall, or during drought.

Large prairie fires pose a serious threat to the sustainability of wildlife habitat and livestock grazing in riparian zones: greater woody vegetation cover and greater forage production increases the amount of fuel present, prolonging the duration of fire in a specific location, resulting in a more severe fire impact. As fires gain momentum, the continuous fuel source available across the landscape further increases the intensity of the fire, especially where fuel is denser, such as in riparian zones (Wowk 2000, Pyne *et al.* 1996). Wildfires burn in an irregular pattern across the landscape, and when fire persists in an area more vegetation is removed and the chance of plant mortality increases as the fire consumes more of the plant's biomass.

1.3 Goals and Objectives

This research was designed to quantify and compare the temporal changes in vegetation cover differences between burned and unburned fire disturbed riparian zones scattered across the landscape. Landsat satellite image data have commonly been applied to study extensive homogenous areas of the landscape, however inconsistent results have been observed at finer scales. By applying Landsat 7 satellite image data in this project to monitor vegetation change within riparian zones, the suitability of Landsat 7 technology application to smaller, more heterogeneous areas within this landscape specifically riparian zones will be tested.

The goal of this research was to assess the temporal change in vegetation cover differences between burned and unburned portions of selected riparian zones across the

landscape in the rangeland of southwestern Alberta since the 1997 Granum Wildfire. To accomplish these goals, the first objective was to identify and map riparian zones in burned and unburned zones using field observations, Landsat 5 & 7 satellite images and aerial photographs. The second objective was to assess post-fire vegetation differences between burned and unburned areas within the riparian zones using field observations, then assess the relationship between field and Landsat 7 data. The third objective was to assess post-fire change in vegetation cover differences between burned and unburned areas of among riparian zones using a temporal analysis of Landsat 7 data.

1.4 Testable Hypotheses

Figure 1.2 illustrates the expected mean vegetation cover differences between burned and unburned areas following fire. Immediately following the fire in the spring of 1998, the expected vegetation difference is hypothesized to be very great, then decrease over time as the plant community returns to a stage similar to pre-fire conditions. The hypotheses to test the validity of the expected deviation for this research project include:

Hypothesis #1	There are some vegetation differences between burned and unburned treatments as observed in the field during 2001 and 2002.
Hypothesis #2	Riparian zones in this landscape may be accurately located in Landsat satellite image data.
Hypothesis #3	Vegetation classes observed in the field can be identified in Landsat satellite image data using various analysis techniques.
Hypothesis #4	Multi-temporal Landsat satellite image data can be used to detect changes in vegetation cover over time.

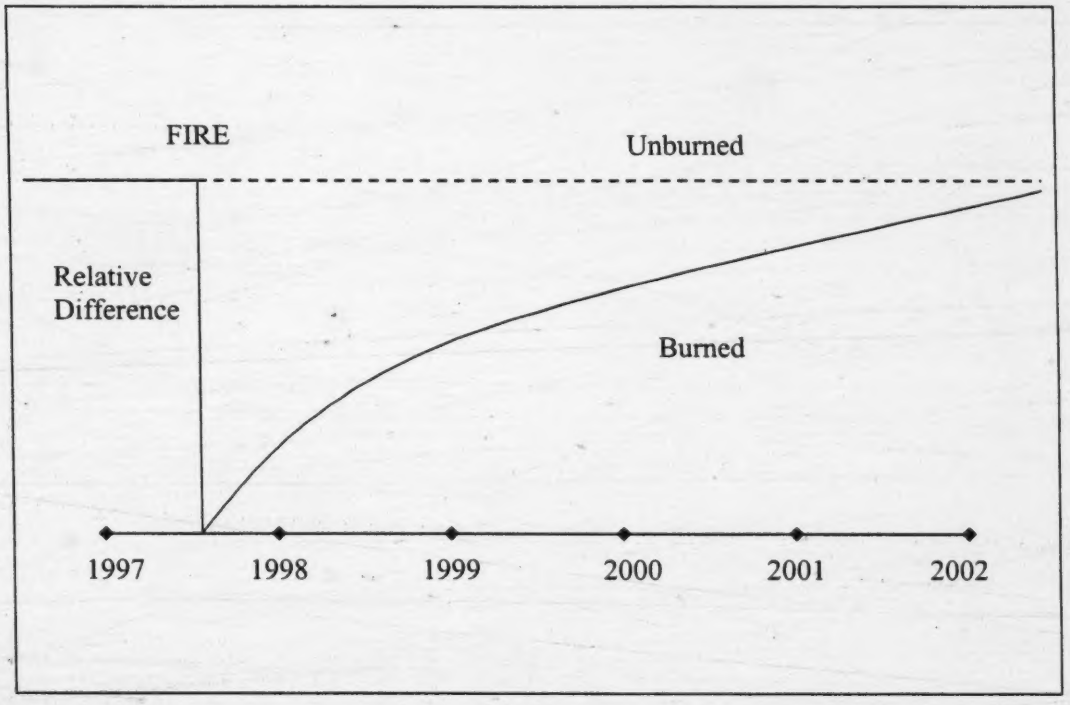

Figure 1.2 Hypothesized mean vegetation differences between mean burned (solid line) and unburned (dotted line) areas following fire.

1.5 Study Area

1.5.1 Physical Description

The study site is located southwest of Granum, Alberta, Canada in the Fescue Prairie ecoregion (Strong and Leggat 1992). The fire occurred on December 14th, 1997 covering a total of 21,600 hectares, approximately 18,000 ha of which was native rangeland (Bork *et al.* 2000, Wowk 2000). The fire ignition and rapid spread were facilitated by a number of factors including the practice of fire suppression resulting in a build up of litter, the unusually warm and dry autumn in 1997, and the development of high winds reaching 70 km/hr shortly after time of ignition. The fire burned northeast from the ignition point at the south end of the Porcupine Hills, across Beaver Creak,

north of Head Smashed in Buffalo Jump, down the east slope of the Porcupine Hills, and was finally extinguished at highway 2 near the town of Granum; travelling a total distance of 31.5 km (Bork *et al.* 2000, Personal Communication with Landowners) (Figure 1.1). Tymstra (1998) estimated the fire intensity to be very high, ranging between 10,000 and 20,000 kW/m. Fire impact included cattle death and loss of valuable stock piled fall/winter biomass for grazing. Additionally, many foothills rough fescue (*Festuca campestris* Rydb.) plants were killed (Bork *et al.* 2000, Willms *et al.* 1993). Considerable amounts of litter were removed, many plants and their root systems were killed, and plant roots and bare soil were frequently exposed. Rough fescue grassland is adapted to frequent fire, not intense fires resulting after years of fire suppression; it was estimated that the last fire in the region occurred over 50 years ago. Intense fires easily damage rough fescue because its growing point is elevated above the soil surface typical of bunchgrasses (Bunting *et al.*. 1998). Recovery of vegetation following the fire may have been delayed because of heavy soil damage and subsequent grazing pressure.

Fescue grasslands are characterized by thin, black chernozemic soils of variable depth (Willms *et al.* 1992, Bork *et al.* 2000). The native vegetation is adapted to conditions of limited and variable nutrient and water supply. In response to these factors, the plants have developed extensive roots systems (Willms *et al.* 1992). Under optimum conditions production reaches 2800 kg/ha with a stocking rate of 0.61 ha/animal (Willms *et al.* 1992). Fescue grasslands of southwestern Alberta are also unique because these areas contained refugia from the most recent glaciation and thus, support a unique diversity of vascular plant and grass species. The dominant range type in the region is the

Festuca-Danthonia plant association (Willoughby and Smith 1996) with the lush plant growth occurring in riparian zones contributing to wildlife and cattle grazing areas.

Large topographic variations, ranging between 760 and 1650 m above sea level (Willms *et al.* 1992, Wowk 2000), produce occasional warm, west winds during the fall and winter that are important to remove snow for wildlife and cattle grazing. The prevalent southwest wind shaped the landscape by creating harsher environments on the southwest facing slopes. According to landowners, strong winds often blow away any litter that forms on these southwest-facing slopes causing decreased plant stature and an overall decrease in percent ground cover. In contrast, the more sheltered northeast facing slopes support denser and taller vegetation as well as a more even layer of litter. These observations apply to most of the area, but are more prominent on hills surrounded by lowlands and towards the shoulder of slopes.

The hilly terrain of the region is difficult to access and thus, is mostly uncultivated (Willms *et al.* 1992) and predominantly used for cattle ranching. This land is shared with a number of wildlife species including bears (*Ursus americanus* and *U. arctos*), elk (*C. elaphus*), white-tailed deer (*O. virginianus*), mule deer (*O. heminionus*), sharp-tailed grouse (*Tympanuchus phasianellus*), and more recently wolves (*Canus lupis*) and cougars (*Puma concolor*). Immediately following the fire, resident elk, deer and grouse were forced to move to neighbouring lands for food and shelter. Anecdotal evidence from the landowners suggests the move resulted in drastic decline in grouse numbers, supposedly due to increased predation and stress in the unfamiliar territory. Presently, elk, deer and grouse again inhabit the area, and their populations are increasing.

1.5.2 Precipitation

The climate normal average precipitation for the Five Mile Creek Weather Station (Figure 1.1) (Nadeau, Personal Communication) is 511.6 mm per year (Figure 1.3). In 1997, 1999, 2000 and 2001, total precipitation was below average, while above average in 1998 and 2002. In 1997, the majority of precipitation fell in May, in 1998 precipitation peaked in May and June, in 2001 April and June and in 2002, peak precipitation was in June with greater amounts of precipitation in May and August (Appendix A).

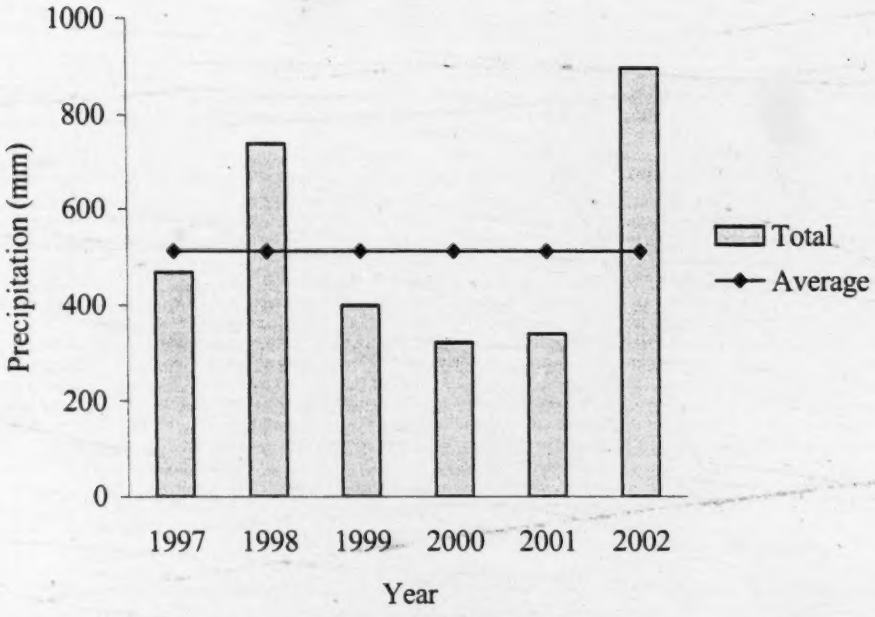

Figure 1.3 Total annual precipitation in relation to the 15-year average recorded for the Five Mile Creek weather monitoring station.

Weather during the summer of 2001 according to landowners was particularly dry and was considered a drought year for the second consecutive year. While, extra snowfall in April and rain in June were sufficient to maintain plant growth throughout the season,

some ranchers suggested the grasses were not as abundant in the summer of 2001 while forbs, especially *Artemesia ludoviciana* Nutt. and *A. frigida* Willd., were in greater abundance. In contrast, in 2002 following a moist spring, the ranchers reported annual forbs to be common.

1.6 Thesis Outline

Relevant literature on the importance of native rangeland and riparian zone ecosystems, the role of fire and grazing in those ecosystems, and the exploration of the use of remote sensing in rangeland and riparian zone studies, are discussed in Chapter 2. Results of the field study comparing burned and unburned treatments in 2001 and 2002 are presented in Chapter 3. Chapter 4 addresses the importance and results of accurately coregistering multiple digital data sets. The suitability of Landsat 7 use with different classification methods to identify vegetation in riparian zones along transects is discussed in chapter 5. Chapter 6 examines the temporal change in vegetation cover differnces between burned and unburned riparian zones across the landscape using Landsat data with various data characterization methods. The final chapter, Chapter 7, summarizes the results found in this research project and suggests alternative research methods.

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Chapter 2 Review of Relevant Literature

2.1 Native Grassland Rangeland

2.1.1 Description of Native Rangeland

The Grassland Ecoprovince, located in southern Alberta, covers approximately 140,000 km^2 (~21.5%) and is characterized by extensive rangeland (Strong and Leggat 1992). Four ecoregions define the Grassland Ecoprovince, the Dry Mixed Grass Prairie in the southeastern corner of the province, the Fescue Grassland to the west, the Mixed Grass Proper between the Dry Mixed Grass Prairie and Fescue Grassland, and the Aspen Parkland to the north (Smith *et al.*, 1995; Strong and Leggat, 1992). Although the total amount of rangeland within each ecoregion varies depending upon agricultural practices, it still comprises an extensive proportion of the land. The Mixed Grass Proper and Dry Mixed Grass Prairie have a combined total of 24 million ha of rangeland (Willms and Jefferson 1993).

Rangeland is composed of range types, which are prototypical communities that the range can potentially support when appropriate disturbance regimes and land management occur, resulting in excellent range condition. However, in reality a diversity of plant communities is found across the landscape based on topo-edapho-microclimatic feature variations. A diagnostic range type has been identified to determine rangeland condition for each range site in each ecoregion. The Dry Mixed Grass Prairie is best characterized by a *Bouteloua-Stipa* range type, Mixed Grass Proper by *Stipa-Agropyron*, Aspen Parkland by *Festuca-Stipa*, and Fescue Grassland by *Festuca-Danthonia* (Strong and Leggat, 1992). The rangeland is also composed of a number of forbs, shrubs and trees including for example, Fabaceae, *Prunus spp.* L., Rosaceae, *Amelanchier alnifolia*

Nutt., *Symphoricarpos occidentalis* Hook., *Artemisia spp.*, Salicaceae and Betulaceae (Smith *et al.*, 1995; Strong and Leggat, 1992).

Topo-edaphic features play an important role in defining plant communities. Topography varies from relatively flat prairie in the Dry Mixed Grass Prairie, to gentle or strong undulations in the Aspen Parkland, and foothills in the Fescue Grassland. Topography influences the amount of insolation reaching the vegetation as well as soil stability, which in turn influence the ability of seral plant communities to establish on the site. Most of the soils in Alberta's Grassland Ecoprovince are well-drained, brown and dark brown, nutrient rich Chernozems with sandy-loam texture developed from glacial tills (Smith *et al.* 1995, Strong and Leggat 1992). The array of soil types, however, ranges from saturated Gleysols in the wetlands, to nutrient poor Luvisols under aspen stands and to Brunisols on eroded surfaces.

The summer precipitation in the Grassland Ecoprovince ranges from 156 mm in the Dry Mixed Grass Prairie to 259 mm in the Aspen Parkland, with the greatest temperatures and longest growing seasonsin southern Alberta (Strong and Leggat, 1992). In the south, moisture is the limiting factor for vegetation growth, whereas further north in the Aspen Parkland and west in the Fescue Grassland, with a precipitation to evaporation ratio approaching 1.0, the shorter growing season is the more prominent limiting factor.

Rangeland covers 30 to 40% of the global land base and is defined as land where the natural potential of vegetation is predominantly grasses, grass-like species and shrubs suitable for grazing and browsing (Tueller 1982, Johnson and Mayeux 1992). The multiple competing land uses found on rangelands include forage for domestic and wild

animals, water, fuel, wildlife shelter, aesthetics and recreation (Tueller 1982, RWRP 2000). Rangeland, characterized by large extent and lower vegetation productivity, is the largest single land use world-wide and thus requires an inexpensive evaluation and management technique to monitor the resource (Tueller 1982).

Sustaining rangeland is a social and ecological process (Huntsinger and Hopkinson 1996). The goals of rangeland management are to ensure sustainable yield and maintain multiple land uses (Tueller 1982). Rangeland in excellent condition is characterized by high levels of native plant species' diversity, productivity and quality, as well as minimal surface soil erosion and the presence of cover for animals (Bork *et al.* 1999a). Active management is necessary to conserve the rangeland landscape because the prairie plant species have evolved in conjunction with fire and grazing disturbances. Grazing is variable across the landscape and forms heterogeneous patches of habitat ranging from heavily used areas with short grass to moderately used areas with a mix of grasses and shrubs, to infrequently used areas where taller, denser shrubs and some trees have developed. Ranching currently represents the primary mechanism for active grazing management of rangeland and also provides a buffer between urban development and natural areas (Huntsinger and Hopkinson 1996). In this buffer, wildlife habitat, watersheds and open spaces are protected while land is still being used for domestic animal production.

2.1.2 Riparian Zones in the Native Rangeland

Riparian zones are the low-lying, moister draws and valleys that are typical of wetlands located within rangeland. Lotic riparian zones are associated with rivers,

streams or drainage ways, and contain a defined channel and floodplain, in which water flows periodically or continually (Brooks *et al.* 1989, RWRP 2000, Naiman *et al.* 1993). A wetland is characterized by hydrophytic or phreatophytic vegetation, undrained hydric, Gleysolic and organic soils, seasonal flooding (Carter 1982), topography (Adams *et al.* 2000) and have a recharge and discharge area. According to Carter (1982) and Cowardin *et al.* (1979) there is no universal definition of a wetland due to their dynamic nature where hydrology, soil and vegetation interact to form the ecosystem. Windel *et al.* (1986) further state that wetland boundaries are hard to discriminate because they are continuous across a moisture gradient and vary with each year. Factors affecting wetland development include the amount of water present, duration and frequency of flooding (Gregory *et al.* 1991). Additionally, nutrient and chemical content of the water, vegetation succession, human activity, climate change and sedimentation also affect wetlands.

The larger the riparian zone the more complex the environmental gradients, ecosystem processes and communities therein. Riparian zones typically have greater vegetation growth than adjacent uplands with more litter and nutrient cycling due to the high amount of moisture present, and therefore play a vital role in ecosystems (RWRP 2000). These functions include 1) soil stabilizing by sediment trapping and bank building and maintenance, 2) capture and beneficial release of water through water storage, aquifer recharge and flow energy dissipation, 3) maintenance of biotic diversity, and 4) provision of primary production (Adams *et al.* 2000, RWRP 2000). Although shrubs and trees are common in riparian zones, and their root systems provide bank stabilization,

while their canopy provides shelter for soils, water and animals, not all riparian zones support trees (RWRP 2000).

The riparian zones in this study cover small sections across a landscape bounded by rangeland and cultivated land. They are therefore the interface between the terrestrial and aquatic ecosystems from the stream channel to the uplands, where sharp environmental gradients, ecological processes and communities occur (Hunt 1985, Naiman *et al.* 1993, Gregory *et al.* 1991). The characteristics of these riparian zones include heterogeneous vegetation, variable topography, and dynamic water flow in the spring and summer. The predominant land use activity in these riparian zones is cattle and wildlife foraging and shelter.

2.1.3 Disturbances in Native Rangeland and Riparian Zones

Maintaining healthy riparian zones across the rangeland is very important to support grazing by livestock and wildlife (Asamoah *et al.* 2003b,in press). Ecosystems are naturally dynamic over time and space in the absence of disturbances (Larsen *et al.* 1998). Following disturbance, native riparian zone communities may succeed into different climaxes because of the post-disturbance plants species composition, seedbank composition and new environmental conditions of the site. For example, exotic species may invade and replace native species resulting in decreased forage productivity, soil stability and fulfillment of complete riparian functions (RWRP 2000, Bailey 1978). In contrast exotic species may be removed and the native vegetation restored. However, overuse of rangeland including riparian zones usually leads to growth of undesirable herbaceous and woody species because preferred species have been eliminated. The

vegetation present in a given year is therefore a snapshot of the effect of weather conditions and site dynamics at that time (TGUCT 1995).

Rangeland and riparian studies typically rely upon a comparison of the current vegetation composition to that without disturbance or treatment, a potential natural vegetation. When this potential composition is reduced in any way, the site is deemed over used; however, reduction of range or riparian use may not cause the vegetation to recover (TGUCT 1995). Change and disturbance are natural components maintaining the rangeland ecosystem. A vegetation community will not always respond in a linear fashion towards a desired composition with improved management. The assumption that a vegetation community needs to meet specific criteria for wildlife habitat also does not consider that each plant community from early seral stage to potential natural vegetation serves different wildlife needs (TGUCT 1995).

Native rangeland and riparian zones have adapted to multiple, highly variable processes including fire, flooding, drought, grazing, erosion, deposition, and soil disturbance. The extent, intensity, location and frequency of these processes form a diversity of vegetation across the landscape (Naiman *et al.* 1993) in different stages of succession, which are important for helping the collective group of communities withstand disturbances (TGUCT 1995). Typically, more diverse communities are resilient and can withstand disturbances, however, continual and frequent disturbances tend to reduce the vigour of long-lived perennials. As a result, species such as weeds, more adapted to frequent disturbance start to increase in numbers and also those plants not grazed due avoidance by livestock (Adams *et al.* 2000). The effect of fire on the differences in vegetation cover across riparian zones in the landscape is the focus of this

thesis, although the vegetation in the study zone is highly dependent upon grazing management.

2.1.4. Effect of Fire on Native Rangeland and Riparian Zones

The response of vegetation cover distribution following fire is site specific (Goodin and Henebry 1997, Larsen *et al.* 1998). Though it may cover a large portion of the landscape, fire does not burn uniformly, thereby creating a patchy landscape. Wildfires typically burn across the surface of the prairie without killing plant root systems. However, according to landowners in the study area, because fire has been routinely suppressed, litter build up provided adequate conditions for a severe fire (Bork *et al.* 2000, Wowk 2000).

Fire is a major environmental factor affecting plant community composition and structure. Fire is beneficial in riparian zones in grasslands because it will remove shrub cover and decadent woody growth to free up resources for more forage growth (Willms *et al.* 1992, Kay 2002, Bailey 1978, Bailey 1986, Vogl 1979). Post-fire vegetation is typically more palatable and nutritious due to the increased nutrients available in the soil, and thus is a preferred forage source. In addition to improving forage quality, fire impedes colonization of non-fire adapted invading plant species, alters soil structure to benefit germination, and reduces insect populations (Vogl 1979), which together promote native plant community recovery. However, immediately following a fire, forage production may decrease as a result of high fire intensity removing too much litter and soil or through plant mortality. A moderate amount of litter is very important to site stability because it protects the soil surface from extreme temperatures, reduces water evaporation and provides for emergency feed (Willms *et al.* 1992). Excessive amounts of

litter on a site reduce forage production, while sparse litter cover allows for increased site erosion. Moreover, dry litter is essential for a fire to burn (Bailey 1986). When large areas are disturbed, the resulting site modifications such as litter loss (Bork *et al.* 2000) may likely lead to reductions in soil moisture, increases in absorption of solar radiation, higher soil temperature, and higher evapo-transpiration due to the darker soil in the disturbed areas (Willms *et al.* 1992, Defossé and Robberecht 1996). Defossé and Robberecht (1996) observed that post-fire plant growth is heavily influenced by the environmental conditions created following the fire and the plant's ability to compete for water, though water is the more prominent factor. Plants in water-limited environments tend to produce less biomass in the season immediately following the fire. In addition, the direct effects of fire on plant tissues, the more stressful post-fire environmental conditions, and short-term competition from annual grass or herbaceous species heavily influence post-fire re-growth of perennial grasses. Competitive fire-sensitive species may become more dominant than fire-tolerant species in the post-fire community (Defossé and Robberecht 1996). According to Bartos *et al.* (1994) forbs outcompeted grasses immediately following fire to form a dense understory cover. To increase the likelihood of community survival and recovery, delaying grazing (Bunting *et al.* 1998) and maintaining high diversity within the community is recommended.

2.2 Remote Sensing of Native Rangeland and Riparian Zones

Rangeland and riparian zone monitoring is necessary for maintaining viable animal populations and properly functioning ecosystems on high quality range. Rangeland and riparian zone condition, is evaluated based upon the presence of exposed

soils, rocks, microphytes, total live vegetation, litter, grasses, forbs, shrubs and trees (Bork *et al.* 1999b). When monitoring the rangeland with remote sensing techniques, the detection of these indicators is important.

Remote sensing entails gathering data where the observer is not in direct contact with the source of the data (Hardy 1982). Application of this form of technology requires the development of a consistent method and means for collecting inventory data for accurate and repeatable comparisons. Remote sensing using satellite data is an efficient means of monitoring land resources because it can be accomplished in inaccessible, remote areas frequently throughout the year. Furthermore, remote sensing improves upon traditional methods of gathering information regarding land resources because it provides an unbiased and accurate portrayal of the entire landscape at a specific location and time in a geographically referenced and storable format. In contrast, traditional field studies only sample a fraction of the landscape (Senseman *et al.* 1996). Thus, remotely sensed data are accessible from an archive and a time analysis can be performed at a later date to observe synoptic land cover changes.

Information needs among remote sensor users are diverse and dynamic (Johannsen and Sanders 1982). In agriculture and range management applications of remote sensing, data need to be near-real time with accurate estimates of land cover and decision making information for planting, range forage production, grazing capacity, inventory and modelling. For wildlife and ecology studies, remotely sensed data is often used for biomass estimation, vegetation mapping in extensive remote areas, and habitat change monitoring for planning and resource management (Estes *et al.* 1982, Johannsen and Sanders 1982).

Each type of remotely sensed data is unique and has specific advantages and disadvantages for use in a project (Hardy 1982). Data specifications need to be assessed to select the most efficient and cost effective data format available. The key to ensuring appropriate information is identified in a satellite image entails selecting a season of monitoring where target data are observable. Weather conditions and satellite cycle may limit when data is accessible, thus restricting the scope of projects (Hardy 1982).

Landsat 7 satellite data are of moderate resolution, designed for application in agriculture, geology, forestry, regional planning, education, mapping and global change (Tables 2.1 & 2.2). Landsat 7 satellite data were selected as an appropriate data source for this landscape-scale project because of the application to agriculture and the repetitive acquisition available for time change analysis. In addition, Landsat 7 data are compatible with the Landsat 5 data used in the previous research projects (Wowk 2000, Speiss and Marciak 1998). The benefit to using the Landsat series data to monitor changes across this landscape is that wildfires are unpredictable (Pyne *et al.* 1996) over time and space, and thus prearranged data collection cannot occur (Turner *et al.* 1997).

Table 2.1 Technical information for Landsat 7 satellite (USGS 1999).

Date of launch	April 15, 1999
Resolution	30 m Scale = 1:1,000,000
Orbit Altitude	705 km
Scene Dimensions	183 km wide and 170 km long
Orbit Cycle	16 day repeat coverage
Number of bandwidths	7 (Table 2.2)
Satellite Angle	Sun-synchronous 98° inclination and descending equatorial crossing time of 10 am
Radiometric Calibration	Subsystem stabilizers keep satellite stabilized within 0.05° or 5%

Table 2.2 Overview of Landsat 7 spectral bands with description of landscape features best observed using each band (Spiess and Marciak 1998, Lillesand and Kiefer 2000).

Band	Bandwidth	Description
1	0.45-0.52 μm (visible blue)	Penetrates water bodies, discriminates soil and vegetation Strong chlorophyll absorption at 0.45 um
2	0.52-0.60 μm (visible green)	Strong chlorophyll reflectance
3	0.63-0.69 μm (visible red)	Strong chlorophyll absorption at 0.67 um
4	0.76-0.90 μm (near-infrared)	0.70-1.3 um is strong vegetation reflectance due to internal structure of plant leaves
5	1.55-1.75 μm (mid-infrared)	Vegetation reflectance influenced by water absorption at 1.4 and 1.9 um
6	10.5-12.5 μm (thermal infrared)	Thermal band with 60 m resolution
7	2.08-2.35 μm (mid-infrared)	Vegetation reflectance influenced by water absorption at 1.4 and 1.9 um

Many factors affect the utility of satellite data. Hyperspectral systems gather solar reflectance values over numerous narrow bandwidths allowing a more precise definition

of spectral pattern for pixel classification. However, a very coarse spatial scale is needed to adequately gather the solar energy required to record information in such narrow bandwidths (Lillesand and Kiefer 2000). In contrast, IKONOS has a very fine spatial resolution of 4 m, which is beneficial for identifying land cover features based on shape. Similar to the Landsat 7 satellite, IKONOS contains four multispectral bands, 0.45 to 0.52 μm, 0.51 to 0.60 μm, 0.63 to 0.70 μm, and 0.76 to 0.80 μm (see Table 2.2 for Landsat 7 equivalent) and one panchromatic band 0.45 to 0.90 μm (Lillesand and Kiefer 2000). The improved spatial resolution, however, requires a larger 11 bit data format, compared to 8 bit for Landsat 7 data, requiring a considerably greater amount of space for storage and manipulation, which prevents data archiving. In addition, IKONOS does not have a mid-range infrared band, such as 5 or 7 (Table 2.2), to increase vegetation discrimination (Lillesand and Kiefer 2000).

Many factors affect the spectral pattern of land cover features such as soil and vegetation. The soil curve is influenced by moisture content, soil texture, surface roughness, presence of iron oxide and organic matter content (Lillesand and Kiefer 1994). The soil reflectance curve is smoother than the vegetation reflectance curve due to the water absorption bands at 1.4, 1.9 and 2.7 μm, thus facilitating differentiation between soil and vegetation. Temporal effects such as vegetation changes throughout the growing season influence spectral patterns of features (Lillesand and Kiefer 1994). Spatial changes in the landscape also influence the spectral pattern of soil and vegetation.

The atmosphere also plays a key role in spectral patterns by attenuating energy and adding to scatter in all instances of remote sensing. Hazy days are not appropriate for satellite image analysis due to atmospheric interference. Seasonal differences in solar

elevation angle affect spectral pattern by changing the intensity and angle of reflectance (Lillesand and Kiefer 1994).

Problems associated with the use of Landsat data in studies include inherent data characteristics such as cloud cover or haze, resolution, manual or automated interpretation, reflectance, spatial distribution of different land cover features, and accuracy of multi-image registration (Estes *et al.* 1982). The complex nature of the distribution of rangeland and riparian plants with respect to phenology, topography, moisture and small size within the landscape creates a highly variable pattern to analyze. Currently, the low resolution prevents extraction of the various pixel sub-components because the average spectral pattern has low separability from pure samples. To improve rangeland and riparian zone assessment, greater differentiation between critical ground cover components is necessary. Success of using Landsat data depends upon the ability to spectrally separate desired land cover classes (Estes *et al.* 1982).

The selection of suitable satellite data bands is essential to differentiate between critical ground cover components for classification of cover type and assessment of rangeland and riparian zone condition. Forbs are currently differentiated from grasses by a stronger green spectral response, while live and dead vegetation are differentiated based on normalized difference vegetation index (NDVI) (Bork *et al.* 1998). Lichens and dead shrubs have intermediate spectral reflectance patterns and have not been absolutely identified, but they could be used as indicators of poor rangeland. The visible spectrum provides a relatively poor indicator of range condition because although the visible spectrum permits us to measure biomass based on spectral reflectance, we are still unable to differentiate between desirable and undesirable range plant species (Major *et al.* 1994).

In contrast, the visible spectrum is an effective indicator of range condition based on soil reflectance properties because exposed soil indicates land degradation (Major *et al.* 1994). Asner (1998) found that plant geometry plays an important role in differentiation based on wavelength dependent variability in tissue optical properties. For example, reflectance for woody species is consistently lower than for grasses in the visible spectrum ($p<0.05$) and higher in the near infrared spectrum ($p<0.05$) because the more complex structure of the leaf increases reflectance in the near infrared (Hobbs and Mooney 1990). However, there is no significant difference between the reflectance patterns of woody and non-woody plant cover in the mid-infrared spectrum because of the high degree of variability caused by water absorption. Similarly, as litter increased, the near infrared spectrum showed a corresponding increase resulting from the denser canopy, and the visible spectrum increased due to the reduced radiation absorption by degrading chlorophyll pigments (Hobbs and Mooney 1990). Multi-temporal data are essential to accurately compare range condition over the key diagnostic intervals during the growing season to account for the influence of changes in phenology on reflectance spectra. For example, grass and shrub growth peaks in June, while forbs are more prominent in July, and dead shrub, rock and exposed soil are more visible in August during senescence and plant chemical changes (Bork *et al.* 1999a). Based on the complexity of the spectral reflectance patterns and their interaction seen in experimental research by Asner (1998) and Bork *et al.* (1999b), an extensive knowledge of spectral relationships is necessary to effectively utilize satellite data.

According to Stohlgren *et al.* (1998), vegetation heterogeneity is representative of the underlying patchiness of the landscape. This patchiness includes exogeneous

variation in nutrient richness, topography, moisture gradients, disturbances by fire, ground squirrels (*Spermophilis richardsonii*), ungulates and other animals. It also includes endogeneous characteristics of seedbank composition and chance establishment, competition, fire and heterogeneous herbivory and temporal variation such as succession, invasion and climate change. The high complexity of the natural vegetation arrangement has created many difficulties associated with separating the spectral reflectance patterns of pixel sub-components into vegetative, edaphic, topographic, and moisture features. However, the increasing refinement of satellite sensors is improving space borne measurement of vegetation spectral reflectance patterns with higher resolution and accuracy. Satellite monitoring of rangeland condition has improved dramatically over the last 30 years to account for the variability of vegetation, topography and moisture present in rangeland. Conversely, long-term studies are necessary to accurately quantify spatial and temporal heterogeneity. Overall, satellite image data have the potential to be an effective tool for monitoring changes in riparian zones because of its multi-temporal availability over an extensive area.

2.2.1 Data Georeferencing

The accuracy of monitoring vegetation changes in riparian zones is dependent upon correct georeferencing of data, which involves aligning all digital data into the same geographic coordinate system. Data may only be analysed when in the same projection to ensure consistent positional accuracy in all digital data. Image rectification, part of the georeferencing process, involves correcting distortions in raw image data to fit the desired projection coordinates. Incorrect registration of data leads to increased error in

site identification (Elmore *et al.* 2000), classification, and subsequent monitoring of vegetation change (Townshend 1992, Elmore *et al.* 2000). These errors provide false information, leading land and resource managers to incorrectly assess landscape condition, and thereby impede proper landscape management.

2.2.2 Data Classification

Use of Landsat 7 provides the large spatial extent and multi-temporal data required for resource inventories, however, without some form of interpretation to separate land cover features into classes, these data are not useful to resource managers. All remote sensing raster data, including Landsat 7 satellite image data, are composed of a number of squares called picture elements, or pixels. Each pixel contains information about the land cover features based on solar energy interaction with the earth's surface and the subsequent reflection from the earth to the satellite sensor (Hurcom and Harrison 1998). The values of these pixels are the data source for all classifications. Ratios and linear transformation of satellite image data typically yield better correlation between vegetation characteristics and spectral classes than individual, original bands (Senseman *et al.* 1996).

Landsat 7 data are a common tool for resource inventories within landscape level studies because the large coverage extent allows the pixel data to be classified using a uniform method (Bonner *et at.* 1982). There are two main types of classifications; a supervised classification where pixels representing a land cover feature are manually selected to create a classification based on information known about the study area, and an unsupervised classification where an algorithm is applied to create the classification

with minimal user input (Lillesand and Kiefer 2000). In an unsupervised classification the spectral classes are determined first and then information such as field data are attached to assess utility of the classification. One benefit of an unsupervised classification is that because the classes are automatically defined, those classes not readily visible in the data may appear.

The maximum likelihood classification, the most popular classification technique for natural resource mapping (Casals-Carrasco *et al.* 2000), is based on pattern recognition to form clusters where the mean and covariance matrices are derived from a sample of pixels from each band in a data set (Baumgardner 1982, Bonner *et at.* 1982). Isoclustering (ISO) and the principle component analysis (PCA) are two methods of clustering data. An ISO involves clustering pixels from the raw bands according to their similarity to the mean vector of a cluster. In contrast, a PCA is a linear combination of the raw band data designed to enhance classification by reducing redundant information, or the dimensionality, of the spectral data (Lillesand and Kiefer 2000, Estes *et al.* 1982). The first component always contains the majority of the spectral variation and all successive components are orthogonal so each component is uncorrelated. The resulting ISO and PCA statistics are then entered into a maximum likelihood classifier with the relevant bands of data where each pixel is assigned to a class based on its statistical similarity to that class (Jensen 1986, Cassals-Carrasco *et al.* 2000, Bonner *et at.* 1982).

Although Estes *et al.* (1982) suggest that limiting an analysis to one or 2 bands results in loss of considerable amount of information, selecting bands that contain relevant information will alleviate this problem. Lillesand and Kiefer (2000) suggest that discrimination of vegetation is enhanced when using one of the mid-infrared bands;

bands 5 or 7. Casals-Carrasco *et al.* (2000) suggest that band selection should be based on low correlation and high covariance to maximize differentiation between classes present in the Landsat 7 image.

The normalized difference vegetation index (NDVI) (Equation 2.1) is the most widely used band ratio for vegetation mapping (Lillesand and Kiefer 2000). The high reflection in band 4 by leaf mesophyll cells and the absorption in band 3 by chlorophyll pigments indicate healthy vegetation allowing the user to distinguish between woody and non-woody vegetation (Table 2.2). When the ratio is high more healthy vegetation is present, while a low ratio indicates a greater presence of exposed soil or dead vegetation. Furthermore, the NDVI is not affected by variations in scene illumination (brightness) because the ratio still conveys spectral characteristics of land cover features. However, the ratio does not account for intensity variations so two features with similar spectral reflectance slopes may appear similar in the ratio.

Equation 2.1 $$NDVI = \left(\frac{\text{band 4 (near-IR) - band 3 (Red)}}{\text{band 4 (near-IR) + band 3 (Red)}} \right)$$

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Chapter 3 Field Analysis

3.1 Introduction

The riparian zones studied and reported in this thesis cover narrow sections across a landscape bounded by pasture and cultivated land. A riparian zone is the interface between terrestrial and aquatic ecosystems from the stream channel to the uplands, where sharp environmental gradients, ecological processes and community changes occur (Hunt 1985, Naiman *et al.* 1993, Gregory *et al.* 1991). The characteristics of these riparian zones include heterogeneous vegetation communities, variable topography, and dynamic water flow in spring and summer. The predominant land use activities in these riparian zones are cattle and wildlife foraging and shelter. Maintaining healthy riparian zones across the rangeland is very important to support grazing by livestock and wildlife (Asamoah *et al.* 2003b, in press).

Native rangeland and riparian zone species have adapted to multiple, highly variable processes including fire, flooding, drought, grazing, erosion, deposition, and soil disturbance. The extent, intensity, location and frequency of these processes form a diverse and stable landscape (Naiman *et al.* 1993). These diverse patches of vegetation in different stages of succession and composition are important for landscape biodiversity, which will help the community withstand disturbances (TGUCT 1995). Typically, diverse communities are more resilient and can withstand disturbances, however, continual and frequent disturbances tend to reduce the vigour of perennials.

In addition to herbivory and drought, fire is another major environmental factor affecting plant community composition and structure (Daubenmire 1968). The response of vegetation cover distribution following fire is site specific (Goodin and Henebry 1997,

Larsen *et al.* 1998). Though it may cover a large portion of the landscape, fire does not burn uniformly, thereby creating a patchy landscape. Wildfires historically burned across the surface of the prairie without killing the plant root systems, however, according to landowners in the study area, because fire has been routinely suppressed, litter build up provided adequate conditions for a severe fire (Bork *et al.* 2000, Wowk 2000). Fire is beneficial in riparian zones and on the rangeland because it will remove shrub cover and decadent growth to free up resources for more forage growth (Willms *et al.* 1992, Kay 2002, Bailey 1978, Bailey 1986, Vogl 1979). Post-fire vegetation is typically more palatable and nutritious due to the increased nutrients available in the soil, and thus is a preferred forage source. In addition to improving forage quality, fire impedes colonization of non-fire adapted invading plant species, alters soil structure to benefit germination, and reduces some insect populations (Vogl 1979), which promote native community recovery. To identify the effect of fire on these plant communities and their subsequent recovery, collection of field data is required.

3.2 Objectives

The objective of this field study was to identify field-based differences between burned and unburned areas within selected post-fire riparian zones. The catena-topo-sequence across riparian, slope and upland vegetation communities at periodic intervals along each experimental riparian zone were mapped to identify vegetation distribution in the field. The hypothesis that there are some vegetation differences between burned and unburned treatments as observed in the field during 2001 and 2002 will be tested based

on a statistical comparison of visual estimation of ground cover along transects between burned and unburned areas.

3.3 Methods

3.3.1 Site Selection

Field sites within the study area were located using Landsat 5 TM satellite image data, a 1:100,000 township map and 1:60,000 aerial photographs. In the September 1997 satellite image taken three months before the fire, all visible riparian zones, represented by a bright red colour in a band 4-3-3 unprocessed false colour composite, crossing the fire boundary were identified (Figure 3.1). These sites were also identified in aerial photographs to confirm the existence of riparian vegetation. Township maps were used to identify the stream names and their significance because only major creeks were identified on the map. Seven sites were selected in total based upon accessibility and orientation. Similar site orientation maintains a more consistent spectral reflectance pattern from each site. All sites were oriented primarily in a north-south direction where landowners granted permission to access the land.

Justice and Townshend (1981) suggested completing image analysis before going into the field to improve the sampling design. In contrast Stohlgren *et al.* (1998) suggests it is better to complete field sampling at peak biomass to more easily identify plants, and also accurately sample distribution and extent of plant cover. Gathering data as soon as possible after image acquisition in the same season is essential. Morever, the delay between image acquisition by the satellite and delivery to the end user is too long to

complete image analysis before field sampling, making field sampling a short-term priority.

Figure 3.1 Pre-fire satellite image data in a band 4-3-3 red-green-blue false colour composite taken in September 1997, where the yellow line represents the fire boundary with yellow arrows pointing to distinguishable riparian zones.

Linear transects within the study sites were used to survey the vegetation in and immediately adjacent to the riparian zones. Data were recorded of the percent foliar cover of plant types according to herbaceous vegetation, woody vegetation and other cover. Data were not collected according to species because these are not distinguishable using satellite image data, and comparable spatial resolutions of data are necessary (Townshend 2000). Foliar classes were used to represent the percent of area such as 0-5%, 5-25%, etc.

This procedure has been used elsewhere (i.e. Senseman *et al.* 1996), and reduces observer variance (Stohlgren *et al.*1998), but may be too broad to detect spatial or temporal changes if species cover is limited (Stohlgren *et al.* 1998). Thus, more general classes were applied including woody, herbaceous and other cover, to avoid problems with small change detection.

3.3.2 Experimental Design of Field Study

The experimental design for this field study was a randomized complete block design with seven sites as replicates (Table 3.1 and Figure 3.2). Site selection was dependent upon the number of creeks crossing the fire boundary. Each creek was set up as a block to account for the variability among riparian zones across the study area. The assumption for each block was that each continuous riparian zone was formed by the same geological event. Thus, vegetation and physical characteristics would be more similar along a continuous riparian zone than two separate riparian zones (RWRP 2000). Pairs of burned and unburned treatments in the riparian zone were considered the experimental unit and were analyzed as paired-sites.

Table 3.1 Experimental design where the treatment is fire, the two levels of treatment are burned and unburned, and the blocks are the seven riparian zone sites.

Source of Variability	Degrees Freedom		
Fire	(f-1)	1	Treatment: Burned and Unburned
Site	(b-1)	6	Block: Sites 1 through 7
Site x Fire	(b-1)(f-1)	6	Error: Interaction
Total	f x b -1	13	Total degrees of freedom

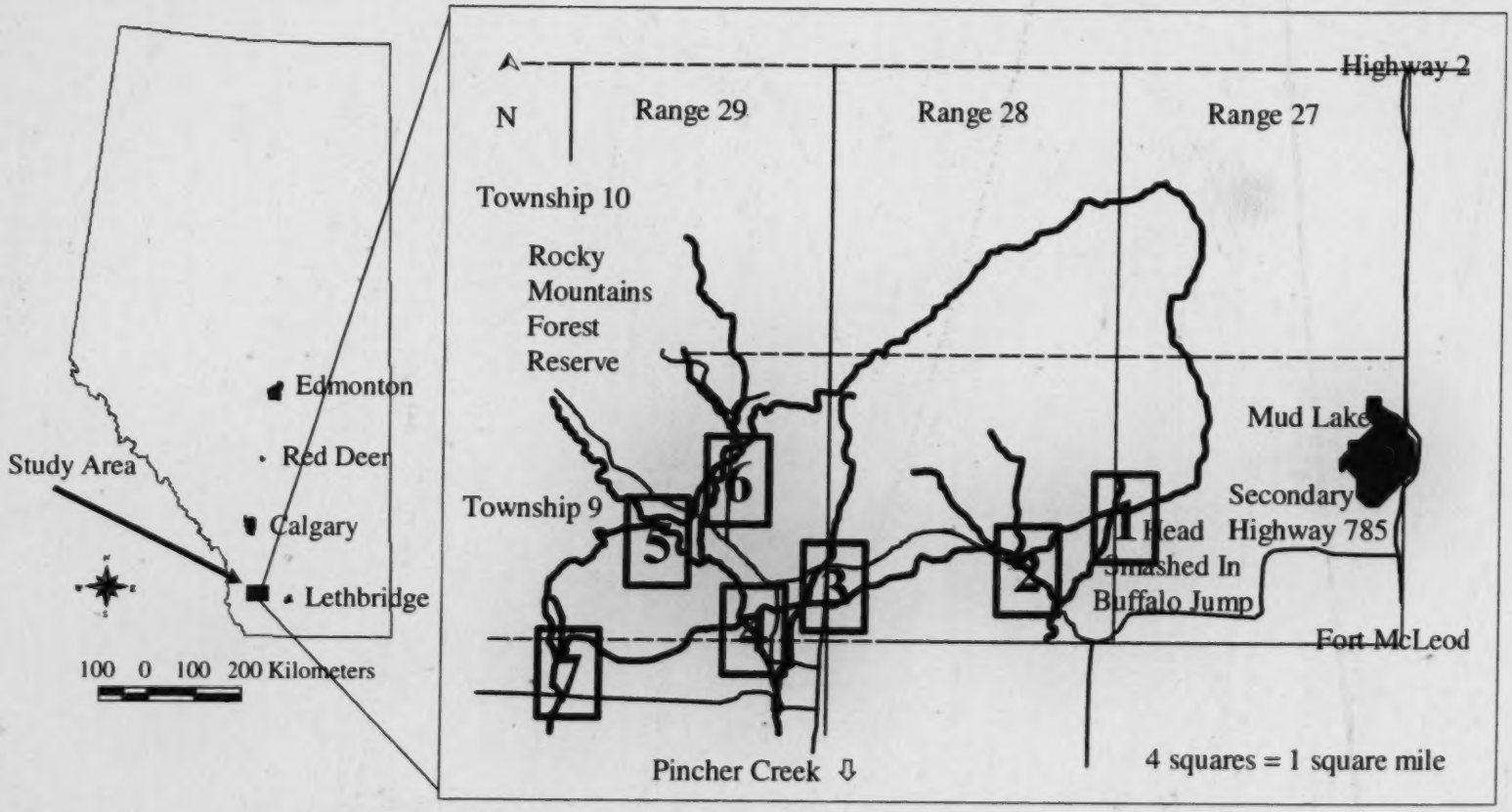


Figure 3.2 Experimental design of 1997 Granum Wildfire study area where the red line represents the fire boundary surrounding native rangeland, the blue lines represent major streams, and the black numbered boxes represent study sites.

3.3.3 2001 Preliminary Field Study

In 2001, field data were collected as point samples using a Garmin 12X GPS unit. A visual assessment was performed across an estimated 25 m by 25 m area surrounding the point for percent cover of herbaceous vegetation, woody vegetation and other cover. Herbaceous vegetation included graminoids and forbs, woody vegetation included shrubs and trees, and other cover included bare soil, litter, standing dead vegetation, water and rocks. Standing dead vegetation and litter were included with bare soil because the spectral pattern of dead vegetation is very dissimilar from live vegetation and more similar to bare soil (Lillesand and Kiefer 2000). The 625 m^2 areas were selected to imitate the land cover that would be found in one TM 25 m^2 x 25 m^2 resampled pixel (Burrough 1998) and the field measurements were taken approximately 100 m apart.

According to Wowk (2000), ocular estimation of large areas is adequate because sampling and averaging small areas produces large variability. Species level studies were inappropriate for a satellite image data analysis because the spectral pattern to differentiate plant species is too coarse. Thus, resolution of the satellite image data is one of the principle factors controlling the sampling area size (Justice and Townshend 1981).

3.3.4 2002 Field Study

In 2002, the field data sampling was completed using 25 m wide band transects approximately 100 m to 250 m apart across the riparian zone to identify both riparian zone vegetation, and vegetation in the adjacent lands to account for variation resulting from mixed pixels (Figure 3.3). The 25 m wide band for the line transects was used to

imitate a line of TM 25 m^2 x 25 m^2 resampled pixels and gather field data representing what land cover features would be observed in each pixel (Burrough 1998).

Across each transect, areas of unique vegetation composition were identified and mapped (Figure 3.3). Boundaries between communities were identified using a Garmin Etrex GPS unit with a combined x-y error of less than 10 m. Direction of error was not calculated because the spatial resolution of the data for this study was 25 m; subpixel error was acceptable for the scope of this project (Senseman *et al.* 1996).

Table 3.2 Key outlining the sequence of steps to determine vegetation class in the field based on percent cover of bare soil and woody vegetation, and average vegetation height.

1a. Greater than or equal to 40% bare soil.	
2a. Litter absent	Bare soil site (BSS)
2b. Litter present	Sparse grass site (SGS)
1b. Less then 40% bare soil.	
3a. Less than 20% shrub cover.	
4a. Less than 15% bare soil.	Lush grass site (LGS)
4b. 15 to 35% bare soil.	Moderate grass site (MGS)
3b. 20-45% shrub cover.	
5a. Shrub height predominantly 20 to 70 cm.	Short shrub-grass site (SSGS)
5b Shrub height mix of 20 to 70 cm and 110 to 70 cm.	Mixed shrub-grass site (MSGS)
5c Shrub height predominantly 110 to 170 cm.	Tall shrub-grass site (TSGS)
3c. Greater than 45% shrub cover.	
6a. Shrub height predominantly 20 to 70 cm.	Short shrub site (SSS)
6b Shrub height mix of 20 to 70 cm and 110 to 70 cm.	Mixed shrub site (MSS)
6c Shrub height predominantly 110 to 170 cm	Tall shrub site (TSS)

Field data were integrated into a GIS in ArcInfo (ESRI 1999) for analysis. The 2002 point coverage was used to create a line transect and then buffered by 12.5 m and

sectioned to recreate the vegetation boundaries in the field. Each polygon representing a unique area of vegetation was assigned a riparian zone identification: upland which included grassland, field and bare ground classes, and lowland, which included the active floodplain, slope and bare soil classes. The channel was classified as part of the active floodplain instead of a unique water class unless the aerial extent was larger than 2 meters.

A new point coverage, representing each unique area of vegetation, was then created within each polygon and the percent land cover attributes were attached. Next, the buffers representing the vegetation community area and the point coverage representing the vegetation field data were spatially joined to analyze the percent cover in each transect and site. Data was originally gathered as percent foliar cover in the field, however, the amount of area sampled varied at each transect and thus, the data needed to be standardized for comparison. Using the area of the vegetation community within the transect buffer, the actual area of land cover was determined and then converted to a percent of the entire transect accordingly. The percent cover of herbaceous vegetation, woody vegetation and other cover within each buffered transect was calculated to assess differences between burned and unburned areas. Vegetation classes (Table 3.2) were derived from a more detailed assessment of the composition of percent cover observed in the field, and from frequency histograms for percent cover of herbaceous vegetation, woody vegetation and other cover (Appendix B). The height component of vegetation classes were distinguished based on predominant or most common average height of cover. Percent cover, cover type and vegetation height all seemed to influence classification techniques in a preliminary assessment of the 2001 data. This percent

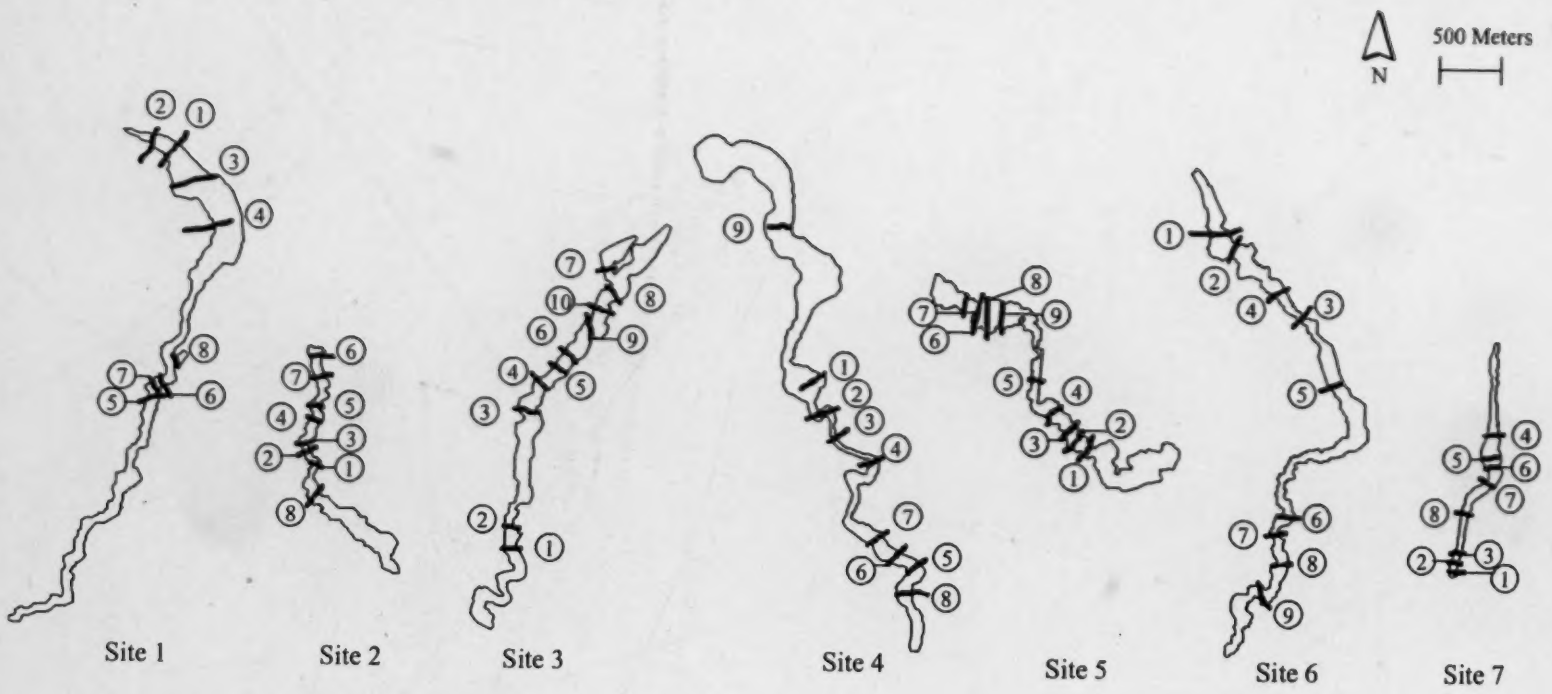


Figure 3.3 Diagram of transects sampled along each site in 2002 (Sites are in numeric order, not geographically oriented across the landscape).

cover information was statistically analyzed using Friedman's Two-Way Non-Parametric ANOVA (SAS Institute, Inc. 1999) to test for differences in percent cover between treatments. Friedman's test was used because the data were categorized into proportions and the distribution did not meet the requirements for normalcy.

3.4 Results

3.4.1 General Site Descriptions

Each of the seven sites was highly variable in topography, ranging from steep narrow draws to wide floodplains. A typical riparian zone cross-section is shown in Figure 3.4. The creek valley bottoms were either incised channels or hummocky floodplains. The depositional banks of the creeks were low lying and covered with a lush mat of herbaceous species. The herbaceous vegetation growing in the very moist parts of the riparian zone was predominately *Carex utriculata* Boott, but also included *Deschampsia caespitosa* (L.) Beauv., *Agrostis stolonifera* L., and *Poa pratensis* L.

On both burned and unburned treatments other common shrubs, including *Eleagnus commutata* Bernh. *ex* Rybd., *Amelanchier alnifolia* (Nutt.) ex M. Roemer, *Prunus virginiana* L., *Potentilla fruticosa* L., and *Crataegus rotundifolia* Moench were present. The *Salix* L. species were often growing adjacent to or in the creek channel. In contrast the *Rosa* L. species and *Symphoriocarpos occidentalis* Hook. were more dominant in the floodplain and on the slopes. *Eleagnus commutata* Bernh. *ex* Rybd. seemed to grow more often on the drier areas of the site.

The area immediately adjacent to the creek or in the lower part of the draw was dominated by emergent wetland herbaceous species and infrequent *Salix* L. species. The

typical height and density of vegetation cover across the creek basin was for shorter, sparser vegetation in the uplands, moderate height and density of vegetation on most slopes, and taller and very dense vegetation cover in the floodplains adjacent to the stream channel. When present, the floodplain was predominantly covered by herbaceous species with some shrubs. The shrub layer often grew in small patches along the creek, extending up to the toe slopes of the surrounding hills. Some shrubs, including taller *Eleagnus commutata* Bernh. *ex* Rybd., *Amelanchier alnifolia* (Nutt.) ex M. Roemer, *Prunus virginiana* L., *and Crataegus rotundifolia* Moench as well as shorter *Potentilla fruticosa* L. extended further up the drier slopes. In general, the upland bordering the riparian zones was either cropland or more often pasture land (Figure 3.4). The vegetation cover on the slopes was variable, consisting of mixed of shrub and herbaceous species.

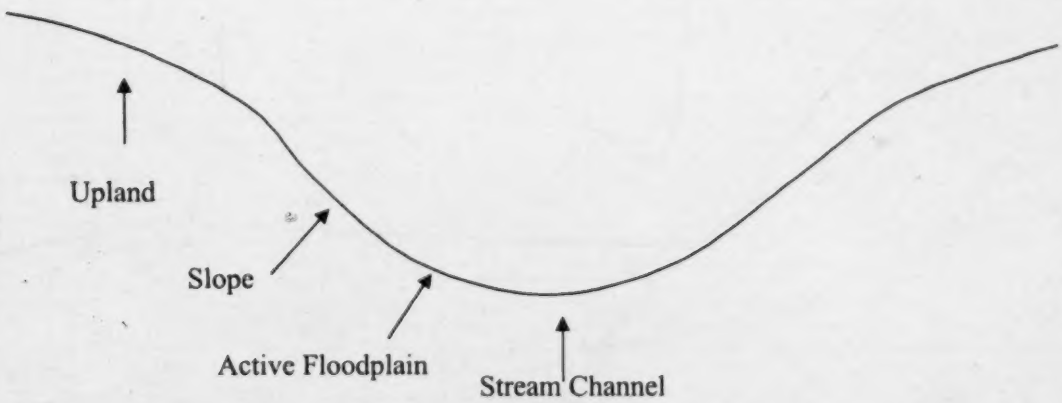


Figure 3.4 Sketch of cross section of a typical riparian zone in the study area.

In the 2001 field study, the burned area was discerned from the unburned by the presence of fallen logs, lower levels of litter and less dead standing wood. The unburned treatment typically contained greater litter and more standing deadwood. However, both burned and unburned sites contained similar plant species composition (Table 3.3) with the exception of *Amelanchier alnifolia* (Nutt.) ex M. Roemer and *Crataegus rotundifolia* Moench, which were more common in unburned sites. Many of the sites contained large amounts of *Bromus inermis* Leyss. and annual forbs on the unexposed slopes, which are not native species. However, on the satellite image this was not visibly discernable from other herbaceous vegetation cover. Characteristics of the vegetation that appeared to differentiate spectrally were a short plant community, a grassland community, an early seral stage shrub community, and a later seral stage shrub community. These observations were used to improve the 2002 field sampling.

All sites had variable grazing management plans, which affected the vegetation composition. The various pastures and grazing pressures affected plant growth. All of the sites were fenced into a number of pastures, and although three of the sites were located on a continuous stretch of one rancher's land, the grazing management in each pasture was not consistent. Also, cattle were present in many pastures season-long but they grazed selectively. Anecdotal evidence from the landowners following the fire indicated that when grazing pressure was removed along the creek areas, tall *Salix* L. species and other taller shrubs were present. In contrast where grazing pressure occurred, the shrub species observed were shorter and included *Symphoriocarpos occidentalis* Hook. and *Rosa* L. species.

In 2001 infrequent ponds were found along half the creeks. However, with the increased precipitation in 2002 (Figure 1.3), all the creeks except site 1 contained flowing water. Despite this absence of flowing water, the ground at site 1 was moister in 2002.

3.4.2 Site 1 Draw on the Calderwood Ranch

Site 1 was a fairly heterogeneous site with respect to vegetation cover and topography, varying from open, grazed herbaceous plant communities to steep, rocky areas with sparse vegetation cover to gradually sloping short-mixed herbaceous and woody vegetation cover. The average width of this site was 121 m and the water flowed from the burned area to the unburned area; an outflow creek. The burned north end of the creek is a steep, narrow rocky draw interspersed with short, herbaceous and woody species. The draw opens into a large meadow over a short distance then narrows again leading into a human constructed dam. Below the dam, the unburned creek narrows and is bordered with moderately sloping, short hills covered with a relatively homogenous short mixed herbaceous and woody vegetation community.

3.4.3 Site 2 Olson Creek on the Dersch Ranch

The topography of site 2 was relatively consistent across the area sampled. The valley bottoms were of varying width but the slopes were moderately steep and of medium height. This site had an average width of 84 m and an outflow creek. Overall, the burned area appeared to have less woody species cover and more herbaceous species cover; predominantly herbaceous. Woody species cover was found mainly on the shoulders of the slopes while forbs were more common on the mid and lower sections of the slopes.

Table 3.3 List of dominant herbaceous and woody plant species across selected burned and unburned riparian zones in 2001.

Site	Treatment	Herbaceous	Woody
1	Burned	*Bromus inermis* Leyss., *Festuca campestris* Rybd., *Achillea millefolium* L., *Solidago missouriensis* Nutt.	*Artemisia frigida* Willd., *Artemisia ludoviciana* Nutt., *Rosa arkansana* Porter, *Symphoriocarpos occidentalis* Hook.
	Unburned	*Carex utriculata* Boott, *Festuca campestris, Koeleria macrantha* (Ledeb.) J.A.Schultes f., *Poa pratensis* L., *Poa sandbergii* Vasey	*Artemisia frigida, Rosa arkansana, Symphoriocarpos occidentalis*
2	Burned	*Carex utriculata, Festuca campestris, Solidago missouriensis*	*Amelanchier alnifolia* (Nutt.) ex M. Roemer, *Artemisia, Eleagnus commutata* L., *Rosa arkansana, Salix* L.., *Symphoriocarpos occidentalis*
	Unburned	*Calamagrostis stricta* (Timm) Koeler, *Carex utriculata, Poa pratensis*	*Amelanchier alnifolia, Artemisia frigida, Eleagnus commutata, Rosa arkansana, Symphoriocarpos occidentalis*
3	Burned	*Carex utriculata, Deschampsia caespitosa* (L.) Beauv.	*Symphoriocarpos occidentalis*
	Unburned	*Carex utriculata, Deschampsia caespitosa, Poa pratensis*	*Symphoriocarpos occidentalis*
4	Burned	*Carex utriculata, Festuca campestris, Poa pratensis, Solidago missouriensis*	*Amelanchier alnifolia, Artemisia spp., Eleagnus commutata, Rosa arkansana, Symphoriocarpos occidentalis*
	Unburned	*Calamagrostis stricta, Carex utriculata, Poa pratensis*	*Amelanchier alnifolia, Artemisia frigida, Eleagnus commutata, Rosa arkansana, Symphoriocarpos occidentalis*
5	Burned	*Bromus inermis, Carex utriculata, Poa pratensis*	*Rosa arkansana, Salix., Symphoriocarpos occidentalis*
	Unburned	*Carex utriculata, Deschampsea caespitosa, Juncus balticus* Willd., *Mentha arvense, Poa pratensis*	*Eleagnus commutata, Salix*
6	Burned	*Bromus inermis, Carex utriculata, Deshcampsia caespitose*	*Eleagnus commutata, Rosa arkansana, Symphoriocarpos occidentalis*
	Unburned	*Carex utriculata, Deschampsia caespitosa, Glyceria grandis, Juncus balticus*	*Salix spp.*
7	Burned	*Carex utriculata*, (heavily grazed)	*Rosa arkansana, Salix, Symphoriocarpos occidentalis*
	Unburned	*Carex utriculata, Calamagrostis stricta, Deschampsia caespitoa, Juncus balticus*	*Eleagnus commutata, Salix*

3.4.4 Site 3 Nine Mile Creek on the Spring Point Hutterite Colony

The topography at the north end of site 3 was composed of moderately steep and high slopes with a relatively large floodplain ranging from 5 to 60 meters wide. Further south the topography changed to gradual short slopes. This site had an average width of 107 m and an outflow creek. The channel of the creek through this site was incised and 1 to 3 meters wide. Overall, the vegetation composition and distribution appear similar between the burned and unburned treatments. However, at the south end of the creek, there were many tall shrubs growing among a lush grass pasture in the riparian zone while in the north and middle sections of this site plant communities were patchy.

In addition to cattle and wild ungulates, ground squirrels (*Spermophilus richardsonii*) played a large role in vegetation composition and distribution in the area. In 2001, a large barren patch caused by ground squirrel grazing was observed in site 3, and these rodents were common throughout the landscape. With the increased precipitation in 2002, there were very few ground squirrels, which according to the landowners, was due to the high water levels that drowned many of them. As a result, the vegetation in 2002 had recovered in site 3 and there was no evidence of large barren patches caused by ground squirrels.

3.4.5 Site 4 Beaver Creek on the Spring Point Hutterite Colony and Newton Ranch

Site 4 was heterogeneous in topography and vegetation cover. The creek channel was bordered by wide floodplains at some sections, and steep hills at others. Herbaceous

vegetation cover mainly dominated the slopes while the valley bottom was generally a herbaceous and woody species mix. This site had an average width of 121 m and an outflow creek. The unburned treatment in site 4 had a number of unvegetated eroding banks. The burned treatment seemed to be composed of a greater cover of tall willows, which may have been a result of grazing management following the fire where cattle were removed (Newton, Personal Communication).

3.4.6 Site 5 Beaver Creek on the Spring Point Hutterite Colony and Price Ranch

Topography between burned and unburned areas in site 5 were very different. The burned treatment contained fewer patches of dead material, but still had occasional solitary dead trees by the creek. In contrast the unburned area contained a considerable amount of decadent woody material. The topography in the burned treatment was high with steeper slopes on the eroding bank and wide floodplains on the depositional bank, whereas the unburned treatment was a very large floodplain with gradual slopes. The average width of this site was 144 m and water flowed form the unburned area to the burned area; an inflow creek; . In both treatments the plant growth of both herbaceous and woody vegetation was denser around the creek channel and floodplain.

3.4.7 Site 6 Five Mile Creek on the Spring Point Hutterite Colony and Nadeau Ranch

The burned and unburned treatments in site 6 had similar topography, but different vegetation cover. In the burned treatment the creek was incised with a relatively gradual sloping floodplain of pastureland on either side. In contrast, the unburned treatment had a wide floodplain, but with high, moderately steep slopes on either side.

This site had an average width of 149 m and an inflow creek. The woody vegetation cover on the burned site was dense, short and patchy, restricted mainly adjacent to the creek channel. The unburned treatment was covered with tall lush grass, interspersed with tall decadent patches of willows growing adjacent to the creek. Vegetation growth in the unburned treatment was considerably taller than in the burned treatment.

3.4.8 Site 7 Creek on the Barr Ranch

Site 7 was a relatively narrow valley surrounded by pasture and fir trees to the north, and pasture to the south. The slopes increased in height towards the south and the site had an average width of 78 m and an outflow creek. Plant communities were small and patchy throughout the site with medium height shrubs in the unburned part and short shrubs in the burned treatment. Both areas were dominated by herbaceous species.

3.4.9 Analysis of 2001 Field Data at Point Samples

In 2001 the field data from each site were compiled and analyzed using Friedman's Two-Way Non-Parametric ANOVA (SAS Institute, Inc. 1999) to test for differences between burned and unburned treatments for the herbaceous vegetation, woody vegetation, and other cover types. The mean percent cover of herbaceous vegetation, woody vegetation and other cover showed no significant difference ($p>0.05$) in treatment effect across all sites (Table 3.4), although there was a trend for more herbaceous vegetation in the unburned treatment, and more woody vegetation and other cover in the burned treatment.

Table 3.4 Mean percent cover of herbaceous vegetation, woody vegetation and other cover for July 2001 in response to burned and unburned treatments following the December 1997 wildfire.

Cover Class	Burned	Unburned	Probability
Herbaceous	43.8	50.9	0.29
Woody	33.1	28.6	0.74
Other	23.2	20.5	0.74

3.4.10 Analysis of 2002 Field Data Along Transects

The randomized complete block design with seven replicates was analyzed using Friedman's Two-Way Non-Parametric ANOVA (SAS Institute, Inc. 1999) to test for differences between treatments for the herbaceous, woody and other cover. The mean percent cover of herbaceous vegetation and other cover showed no significant different

Table 3.5 Mean percent cover of herbaceous vegetation, woody vegetation and other cover for July 2002 in response to burned and unburned treatments following the December 1997 wildfire.

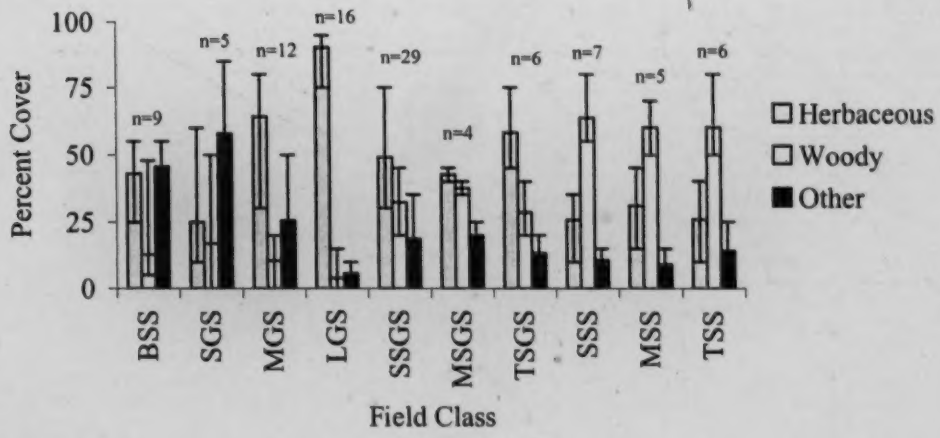
Cover Class	Burned	Unburned	Probability
Herbaceous	60.0	68.8	0.74
Woody	26.0*	20.3*	0.05
Other	14.0	16.2	0.74

*Indicates significant difference ($p>0.05$) between treatment effects.

($p>0.05$) between treatments, while the woody vegetation showed a significantly higher percent cover on the burned treatment (Table 3.5). Similar to 2001, the percent cover of

herbaceous vegetation was higher in the unburned treatment while the woody vegetation was greater in the burned treatment. In 2002, though, the percent cover of other cover was greater in the unburned treatment.

2001



2002

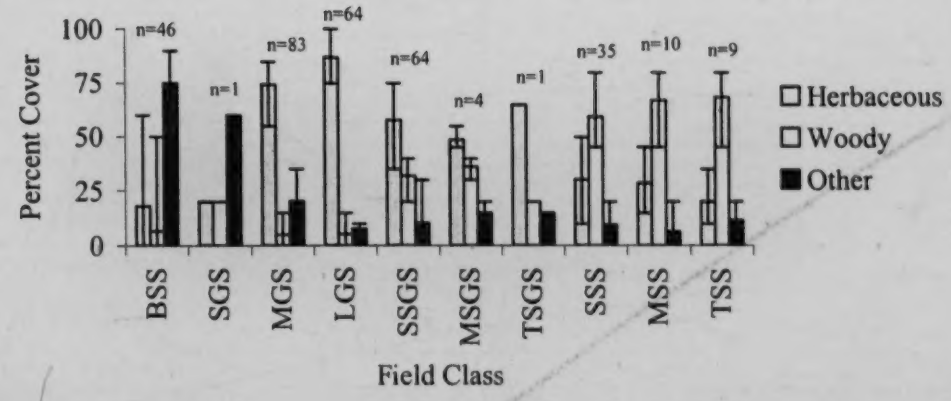
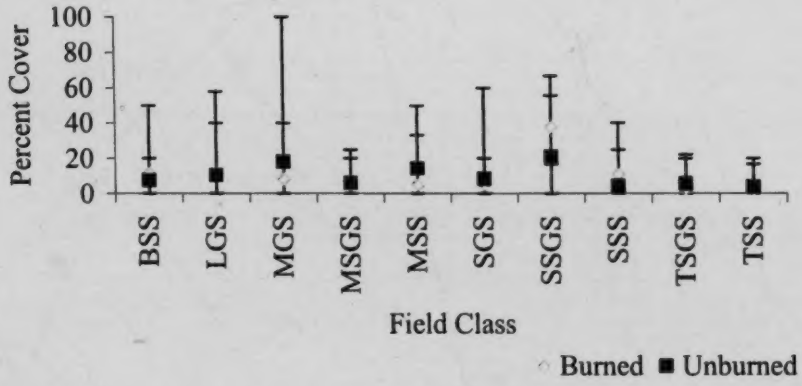


Figure 3.5 Distribution of average percent cover of herbaceous vegetation, woody vegetation, and other cover within each field class (see Table 3.2) in 2001 (above) and 2002 (below) where the y-error bars represent the minimum and maximum percent cover.

2001



2002

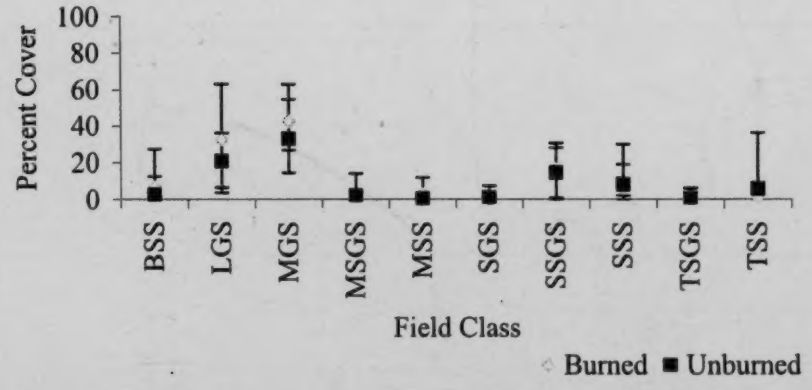


Figure 3.6 Mean percent cover of each field class (see Table 3.2) across the point and transect buffers for the burned and unburned treatments in 2001 (above) and 2002 (below), where the y-error bars show the minimum and maximum range of percent cover within each field class.

Given that data collection methods were different between 2001 and 2002, absolute change in cover cannot be compared. However, from 2001 to 2002 within in the burned treatment, the percent cover of herbaceous vegetation increased by 2%, woody vegetation increased by 4% and other cover decreased by 6%.

Of a total of 99 field sample sites in 2001 and 317 in 2002, 10 detailed classes were identified (Table 3.2). Figure 3.5 displays the average percent cover of herbaceous vegetation, woody vegetation and other cover in each of the field classes for 2001 and 2002. There are four main patterns of cover type combinations including high percent cover of other cover types for the bare soil and sparse grass sites, high percent cover of herbaceous vegetation with low percent cover of woody vegetation for the moderate and lush grass sites, moderate percent cover of herbaceous vegetation with moderate percent cover of woody vegetation for the grass-shrub sites, and low percent cover of herbaceous vegetation with high percent cover woody vegetation for the shrub sites.

Using the 10 field classes derived to assess vegetation differences between burned and unburned areas, figure 3.6 illustrates the mean percent cover of the 10 classes across the entire set of field transects for 2001 and 2002. In both years, field data results did not show any significant difference between burned and unburned treatment effects in any field class. However, from 2001 to 2002 in the burned treatment the average percent cover of six classes, SGS, MGS, LGS, TSGS, MSS, and TSS increased, the percent cover of BSS and MSGS remained similar, while the percent cover of SSS and SSGS, decreased in the burned treatment. In addition, the range of percent cover for all field classes across the landscape decreased from 2001 to 2002.

3.5 Discussion

A high degree of variability across the landscape resulted, likely from different land management practices, topography, wildlife use of the area and pre-fire conditions of the plant communities examined. The riparian zones of the seven sites were very

heterogeneous within site and between sites. Although the vegetation composition was relatively similar in all sites, the percent cover varied between sites.

Field data from 2001 showed no difference in percent vegetation cover between the burned and unburned treatments. This result suggests that the vegetation was similar at this point following the fire based on percent cover of general field classes (herbaceous vegetation, woody vegetation and other cover) relative to detailed classes examined (Table 3.2). The expected dominant vegetation cover following fire would be herbaceous because the fire should have removed a considerable amount of woody species (Bailey 1986). In the 2001 data, herbaceous vegetation was dominant but woody vegetation cover was also very high. Field data from 2002 showed a significantly greater amount of woody vegetation cover in the burned treatment, but no significant difference in the percent cover of herbaceous vegetation and other cover, nor a significant difference in the more detailed field classes (Table 3.2). However, the percent cover of both herbaceous vegetation and other cover tended to be greater (thought not significant) in the unburned treatment.

The observation of a greater (thought not significant) amount of woody vegetation cover in the burned treatment compared to the unburned treatment could be a post-fire response of woody vegetation. Woody species in the study area such as *Syphoriocarpos occidentalis* Hook. and *Rosa* L. species respond to fire by suckering, thus increasing their foliar cover but not necessarily their stature, which explains the increase in percent cover of woody vegetation (Anderson and Bailey 1979, Archibald *et al.* 1998). Furthermore, because cattle do not prefer either of these two woody plant species (Bailey and Anderson 1990), grazing does not hinder woody plant spread to grass dominated areas.

Anecdotal evidence from the landowners indicated that although the pressure of cattle grazing was temporarily removed from the study area to promote rangeland condition recovery following the fire, elk grazing on the study area was high in the burned treatment immediately following the fire, which is supported by other research (Peck and Peek 1991, Bailey 1986, Vogl 1979). This increased grazing pressure from the elk may have reduced the vigour of the herbaceous vegetation in the burned treatment, thus providing the opportunity for woody vegetation spread by removing competition.

3.6 Conclusions

In Figure 1.2 the expected difference between the burned and unburned areas in 2001 and 2002 was low. In 2001, there were no differences in the percent of herbaceous and woody vegetation cover. There was a significant difference in percent cover of woody vegetation in 2002. The analysis of more detailed field classes along transects resulted in no significant differences in vegetation between the burned and unburned areas in 2001 and 2002. Based on these results, there was very little difference between the burned and unburned vegetation based on field observations four and five years after the fire (Hypothesis #1) suggesting recovery had occurred.

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Chapter 4 Image Registration and Site Delineation

4.1 Introduction

Satellite image data are collected in a grid pattern where the value of a pixel (grid cell or picture element) is the area weighted average of the solar radiation received (Burrough 1998). Given that satellite sensors vary in cell resolution, the pixel size limits the spectral resolution as cell size decreases. In contrast, field measurements are sub-pixel and detect variation within pixels. To combine these data sets, areas must be compatible for spectral analysis (Burrough 1998). Furthermore, for multi-temporal data sets, the satellite sensor should be the same or similar, and anniversary dates should be used to decrease variability caused by differences in the sun angle and season (Lillesand and Kiefer 2000).

Monitoring vegetation changes in riparian zones is dependent upon correct georeferencing of data, which involves aligning all digital data into the same geographic coordinate system. Accurate spatial registration within a quarter or half a pixel is essential for accuracy (Lillesand and Kiefer 2000). Image rectification, part of the georeferencing process, involves correcting distortions in image data to fit the desired projection coordinates. Once all digital data are precisely aligned, multi-temporal analysis of the data may proceed. Incorrect registration of data leads to increased error in site identification (Elmore *et al.* 2000), classification, and vegetation change monitoring (Townshend *et al.* 1992, Elmore *et al.* 2000, Smith and Atkinson 2001). These errors provide false information, leading land and resource managers to incorrectly assess landscape condition, and thereby impede proper landscape management.

4.2 Objectives

The objectives of the image registration were to first, register all digital data to within sub-pixel error. The second objective was to confirm registration of the digital data using field collected GPS points, false colour composites composed of multi year image data, and a digital elevation model (DEM) stream network. The third objective was to create a polygon coverage from the aerial photographs delineating the riparian zones to isolate the pixels representing each site within each satellite image. The fourth objective was to identify problems registering data of different spatial resolution. The hypothesis that riparian zones in this landscape may be accurately located in Landsat satellite image data will be tested based on a statistical comparison of error between the satellite derived riparian zones and the actual field measured riparian zones.

4.3 Methods

4.3.1 Data Acquisition

To complete the assessment of changes in vegetation cover differences between burned and unburned areas, Landsat image data for 1997, 1998, 2001 and 2002 were acquired (Table 4.1). Multi-temporal Landsat satellite image data, bands 3, 4 and 5, were acquired for September 1997 (pre-fire) from the Conservation and Development Branch of Alberta Agriculture, Food and Rural Development, for May 1998 (post-fire) from previous research by Wowk (2000), and purchased for July 2001 and July 2002 (four and five years after the fire). A pre-existing digital elevation model (DEM) was acquired from the Conservation and Development Branch of Alberta Agriculture, Food and Rural Development. In addition, 1:60,000 black and white aerial photographs of the area from

Table 4.1 Data Description and Sources

Data	Date	Data Source Description	Data or Bands Available	Spatial Resolution	Data Ownership
DEM	-na-	-na-	Elevation	100 m resampled to 25m	Conservation and Development Branch of Alberta Agriculture, Food and Rural Development
Air Photos	August 9, 1996	Roll # AS4708 Print # 136, 138 and 142	Black and White	1:60,000 scanned at 1000 dpi	Department of Renewable Resources, University of Alberta
Township Map	-na-	-na-	Township lines Stream names Geographic details	1:100,000	Conservation and Development Branch of Alberta Agriculture, Food and Rural Development
Satellite Images	September 22, 1997	Landsat 5 TM Track 41 and Frame 25	3, 4, 5	30 m resampled to 25 m	Conservation and Development Branch of Alberta Agriculture, Food and Rural Development
	May 20, 1998	Landsat 5 TM Track 41 and Frame 25	3, 4, 5	30 m resampled to 25 m	Department of Renewable Resources, University of Alberta
	July 7, 2001	Landsat 7 ETM Track 41 and Frame 25	1 to 7 and Pan-chromatic	30 m resampled to 25 m	Department of Renewable Resources, University of Alberta
	July 10, 2002	Landsat 7 ETM Track 41 and Frame 25	1 to 7 and Pan-chromatic	30 m resampled to 25 m	Department of Renewable Resources, University of Alberta

1996 were acquired by Wowk (2000). These data were then collectively used to analyze the effect of the fire on the vegetation.

Originally, satellite image data from the month of May each year were to be used to maintain consistent seasonal monitoring with research completed by Wowk (2000). Wowk (2000) selected a May image because the spectral differences between land cover classes would be greatest during the spring. However, cloudy weather conditions prevented the acquisition of haze free images until July in both 2001 and 2002. The Conservation and Development Branch of Alberta Agriculture, Food and Rural Development selected an image from September in 1997 immediately before the burn but still during the growing season to maximally differentiate between woody and herbaceous vegetation cover classes (Speiss and Marciak 1998). The precipitation record indicates that a considerable amount of precipitation fell prior to each of the dates when satellite image data were recorded (Appendix A).

Digital analyses were completed using a geographic information system in ARCInfo version 8.0 (ESRI 1999a) and ArcView version 3.2 (ESRI 1999b). These software programs were selected because they were compatible with previous data analysis studies and the software packages are widely used. All data management and analysis were completed in the Spatial Information Systems Laboratory in the Department of Renewable Resources at the University of Alberta, Edmonton, Canada.

4.3.2 Registration of Digital Data

All digital data collected before 2001 were in the projection Universal Transverse Mercator (UTM) 1927 North American Datum (nad27) zone 12, while digital data

gathered in 2001 and 2002 were in projection UTM nad83 zone 12. The studies by Spiess and Marciak (1998) and Wowk (2000) were in UTM nad27 zone 12 projection to maintain database consistency to existing base map and soils data owned by the Government of Alberta, which were registered in UTM nad27 (Speiss and Marciak 1998).

The first task was to co-register all digital data into the same projection. The 1997 satellite image and the digital elevation model data were projected into UTM nad83 zone 12 and compared to the GPS coordinates gathered at road intersections in the study area in the same projection. The 1997 satellite image data did not register correctly with the DEM or the GPS points; the 1997 image was shifted 100 meters southeast. To ensure accurate coregistration of all the data sets, the 1997 image was rectified using the field GPS points. The 1997 image was rectified using the ArcInfo command *Warp* (rotates, changes resolution and changes projection at same time) with an overall root mean square error (RMSE) less than 0.5 (less than half a pixel) in the x and y direction, maintaining sub-pixel error for individual control points. Using only a few points for registration of the satellite image data was acceptable because of the low distortion associated with the affine transformation. An affine transformation uses a first order polynomial equation to mathematically fit the flat plane of the satellite image data to the control points in the forward direction. The DEM generated stream coverage was then overlaid onto the registered 1997 image to confirm accuracy by visual assessment of stream placement. All subsequent digital data were registered to the 1997 image with an overall x and y RMSE of less than half a pixel and sub-pixel error for individual control points. To confirm the accuracy of the coregistration of the remaining satellite images data sets to the 1997

registered image data, false colour composites were created mixing the 1997 registered satellite image with the coregistered images. These false colour composites were visually assessed for agreement of edges based on colour composition.

An affine transformation for satellite image data rectification was appropriate because the higher altitude and smaller instantaneous field of view of the satellite results in low image distortion. Thus, the image does not need correction for polynomial warping, but rather only a rotation and shift to align the data. RMSE was used in this study to determine an appropriate level of location error for registration. Although RMSE assesses accuracy, it provides no information regarding directional variability of error (Owens and McConville 1996). RMSE was the only available statistic to calculate positional accuracy, and is suitable according to Lillesand and Kiefer (2000). Using GPS points for the georeferencing dataset was acceptable because the points were recorded at distinct edges and the georeferencing tested with a DEM (Elmore *et al.* 2000, Smith and Atkinson 2001, Gao 2002). Visual confirmation of image edge agreement using false colour composites is common and an accurate assessment of georeferencing (Lillesand and Kiefer 2000).

Although it is important to spectrally calibrate all images using temporally invariant surfaces, such as deep, clear bodies of water, to standardize the images for spectral analysis (Elmore *et al.* 2000), none of the satellite image data covering the study area contained such a feature. Thus, atmosphere, solar elevation and azimuth will play some role in the observed difference in spectral pattern among sampling times. To account for these spectral deviations, the analysis will compare differences between spectral patterns of burned and unburned treatments within one image, and then compare

the change in this difference between image dates. All image data selected contained little to no haze or cloud cover resulting in a clear image with little visible atmospheric effect. Burned and unburned areas were uniformly affected by these factors within a sampling date.

To maintain a consistent procedure in this study, bilinear interpolation was used to resample all satellite data images that required registration. Bilinear interpolation is appropriate for continuous smooth surfaces because it uses the values from the four surrounding input cell centres to create a distance weighted mean value for the new output cell value (ESRI 1999). The 1997 image was originally resampled using a bilinear interpolation to maximize differentiation of vegetation boundaries (Spiess and Marciak 1998). When correcting the registration of the 1997 image data, nearest neighbour sampling was used to maintain the pixel values close to their original value (ESRI 1999). Wowk (2000) also resampled the 1998 satellite image data using a bilinear interpolation. Thus, these three studies had compatible data sets, to compare differences between treatments in each image. Direct image to image comparison is not recommended because of variable external influences on spectral values within each image, such as haze.

The 1000 dpi scanned aerial photographs were also registered to the UTM nad83 zone 12 projected 1997 image. However, the high distortion occurring in the aerial photographs, from varying angle of photography and effect of the large instantaneous field of view, required a higher order polynomial transformation to reduce the RMSE. Although a higher order polynomial was used for the transformation, the RMSE remained unacceptably high; greater than 10 meters error. To reduce this error, the

individual riparian zones were clipped out of the aerial photographs to allow for an affine transformation with acceptable x and y direction RMSE of less than one meter. However, a second order polynomial transformation was required for site 7 because of the high amount of distortion resulting from the large angle off nadir between the camera and site. Resampling of the aerial photographs was done using nearest neighbour sampling to retain values close to original for a more accurate differentiation of riparian zone vegetation. After co-registration of the aerial photographs, a false colour composite was again created with the 1997 satellite image data to test for agreement of co-registration. The site 6 aerial photograph was observed to not be in registration with the satellite image and GPS data sets and thus, was shifted to correct the alignment; *xshift* –30 m and *yshift* –10 m.

4.3.3 Shift to Correct Alignment Between Image and Grid Raster Data

ArcView 3.2 (ESRI 1999b) and ArcInfo 8.0 (ESRI 1999a) read and display image and grid raster data differently. The point of geographic reference for image raster data is typically based on the upper left-hand corner of the cell, while for grid raster data it is based on the centre of the cell. In ArcView, an image and a grid in the same projection will not display in perfect alignment, but instead the grid will be shifted up and to the left by half a pixel. ArcInfo, in contrast, will align both formats of data properly because they represent the same area, though the point of positional reference is different. Because an image was used in ArcView to gather control points for image rectification, the resulting grid was expected to be shifted half a pixel down and half a pixel right. To compensate for this shift, the grid data in ArcInfo was corrected using the *project* command where the

projection was altered by adding an *xshift* of 12.5 m and a *yshift* of -12.5 m (Table 4.2). This correction ultimately brought the grid data into alignment with the vector and raster data used in this project.

Table 4.2 Projection information to compensate for shift between image and grid raster data between ArcView (ESRI 1999b) and ArcInfo (ESRI 1999a).

Year	1997	1998	2001	2002
Projection	UTM	UTM	UTM	UTM
Datum	Nad83	Nad83	Nad83	Nad83
Zone	12	12	12	12
Units	Meters	Meters	Meters	Meters
Xshift	+12.5	+12.5	+12.5	+12.5
Yshift	-12.5	-12.5	-12.5	-12.5

4.3.4 Delineation of Study Sites from Aerial Photographs

Riparian zone shape files were created in ArcView 3.2 (ESRI 1999b) with *Add a new theme* and delineated by tracing a polygon around the riparian zone vegetation communities and excluding the cropland and upland grassland areas. Once all the polygons composing one site were complete the command *Dissolve* was applied to dissolve the boundaries within one site to create one polygon. The delineated riparian zone polygons were then overlaid on top of all the satellite images to confirm coregistration and inclusion of the pixels representing the riparian zone in the satellite image. Given that the command *Gridclip* did not include pixels that straddled the polygon perimeter, the dissolved site polygon was extended by one pixel (25 m) using the *Buffer* command to completely clip all pixels identified in the aerial photograph as riparian zone vegetation. Each buffered riparian zone was then used to clip the streams with *Gridclip* out of the grid format of the satellite image data for further analysis.

To test the accuracy of the final aerial photograph delineation, an AML (arc macro language) script was written by Pelletier (2003) to find the distance between the digitized riparian boundaries and those boundaries observed in the field. Transect polygons at each site created from the field data were intersected with the delineated riparian boundary. Next, using the riparian identification attributes of the transect polygons, the distance from the edge of the outermost riparian community polygon to the delineated riparian zone boundary was calculated. These absolute numbers were then averaged for each site and for the combined sites to determine approximate accuracy of riparian boundary delineation with the field data.

4.4 Results and Discussion

4.4.1 Digital Image Registration

All satellite image data were registered to the 1997 satellite image to within half a pixel distance error (Table 4.3). Each control point used had sub-pixel error in both the x and y axis direction. Figures 4.1 through 4.4 display the colour composites of the geo-referenced Landsat 7 satellite image data using a band 4-3-3 combination (see Table 2.2) with the fire boundary outlined in yellow. The darker red to the west is coniferous forest, the brighter red is dense deciduous vegetation, the darker cyan colour is upland range, and the bright white and blue are cultivated lands.

Table 4.3 Overall registration error for satellite images reported in fraction of a pixel, where the resolution of one pixel is 25m.

	Year			
	1997	1998	2001	2002
X-RMSE	0.47	0.38	0.46	0.37
Y-RMSE	0.45	0.42	0.41	0.43

All aerial photographs were registered to the 1997 satellite image to within half a pixel distance error (Table 4.4). Each control point used had sub-pixel error in both the x and y axis direction. These errors are minor in relation to the average riparian zone width described in section 3.4.

Table 4.4 Overall registration error for aerial photographs reported in fraction of a pixel, where the resolution of one pixel is 1m.

	Site and Aerial Photograph Number						
	Site 1 #142	Site 2 #142	Site 3 #138	Site 4 #138	Site 5 #138	Site 6 #138	Site 7 #136
X-RMSE	0.23	0.22	0.30	0.41	0.43	0.48	0.38
Y-RMSE	0.33	0.39	0.38	0.27	0.37	0.34	0.47

Figure 4.1 Georeferenced Landsat 5 satellite image data from September 1997 in a band 4-3-3 red-green-blue colour composite.

Figure 4.2 Georeferenced Landsat 5 satellite image data from May 1998 in a band 4-3-3 red-green-blue colour composite.

Figure 4.3 Georeferenced Landsat 7 satellite image data from July 2001 in a band 4-3-3 red-green-blue colour composite.

Figure 4.4 Georeferenced Landsat 7 satellite image data from July 2002 in a band 4-3-3 red-green-blue colour composite.

4.4.2 Aerial Photograph Delineation Accuracy

The average distance error between the delineated (i.e. mapped) riparian boundary and the field data for the seven combined sites was 18 m (16% of average riparian zone width) with a site average error ranging from 12 m to 25 m (Figures 4.5, 4.6 and 4.7). The greatest delineation error was observed in site 4 with a maximum error of 99 m and site 6 with a maximum error of 90 m. In both of these sites, the delineation error was a result of a gradually sloping, lush, herbaceous community being misinterpreted as a continuation of a riparian meadow.

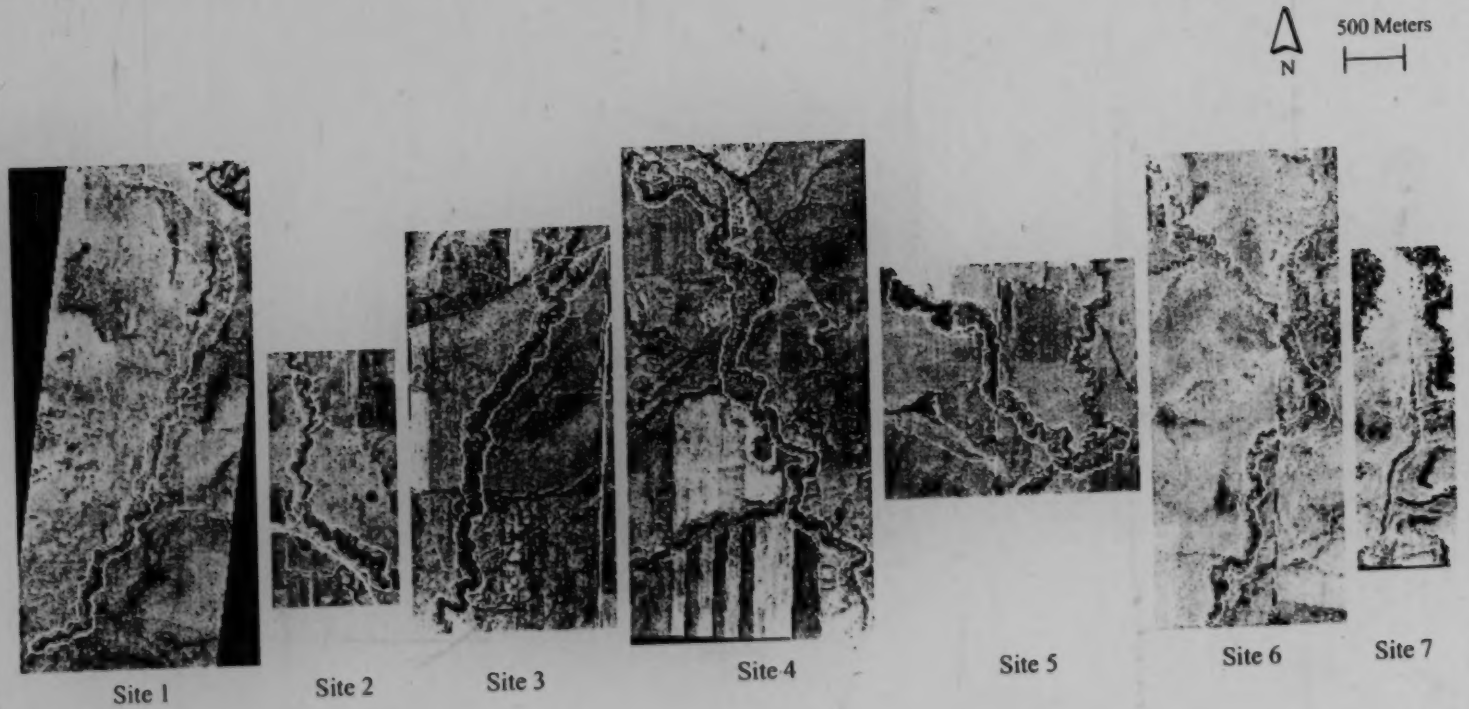

Figure 4.5 1996 georeferenced 1:60,000 aerial photographs with buffer of riparian zone sites outlined in white (Sites are in numeric order, not geographically oriented across the landscape).

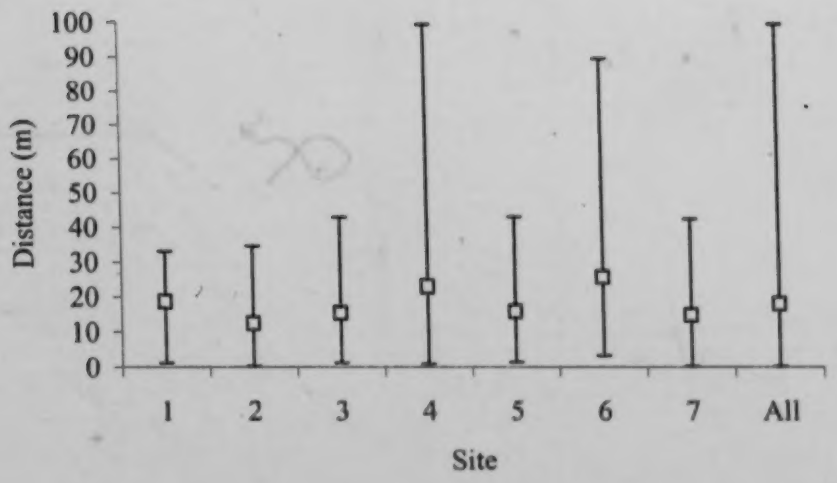

Figure 4.6 Average distance error between mapped and actual riparian zone boundary, where the y-axis error bars represent the minimum and maximum distance observed at each site.

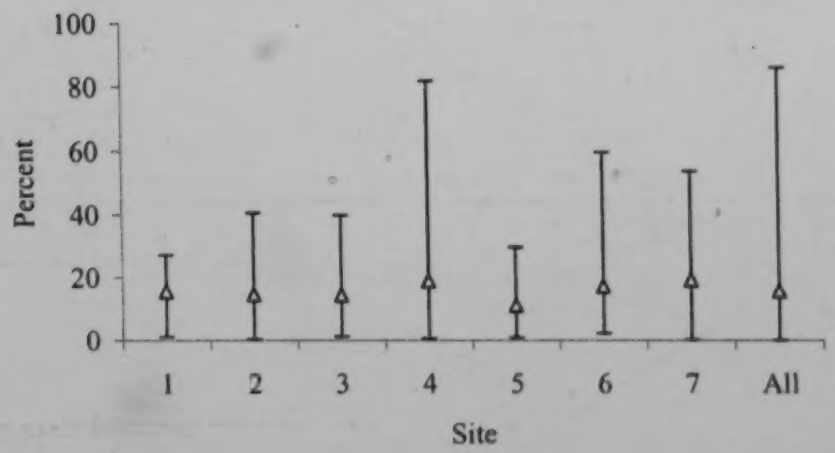

Figure 4.7 Percent average error between mapped and actual riparian zone boundary, where the y-axis error bars represent the minimum and maximum percent distance observed at each site.

According to Grosse and Alder (1996), although aerial photographs contain detailed information, as the map scale decreases distance errors resulting from interpretation increase. Accuracy can be increased where distinct boundaries are present, but often boundary delineation in areas of transition is subjective. Furthermore, Congalton (1991) stated that although photo interpretation is often assumed to be 100% correct, it is often a source of large error. Riparian boundaries in the 1:60,000 aerial photographs taken in 1996 were not distinct and required user interpretation. However, the errors introduced by delineating riparian zone boundaries using aerial photographs were subpixel and they were used with lower resolution satellite image data. That is, these errors were minimal relative to those introduced by the satellite imagery, which had lower spatial resolution than the aerial photography. Thus, interpretation of the satellite image was not impeded (Lodin and Skea 1996) by using aerial photographs to identify riparian zones.

Large variation in distance error between the delineated riparian boundaries and the field data was expected because of the lack of distinct boundaries in the aerial photographs between vegetation communities (Grosse and Alder 1996). Average distance error of the site delineation from the aerial photographs to the field observations was less than 25 m meaning that the error was sub-pixel and should not affect the data analysis because the edge of the upland is part of the riparian zone. Given that the riparian zone vegetation is heterogeneous and bordered by upland cropland and grassland, the pixels observed in this study will inevitably be mixed. As a result, the sampling area needed to be extended to include samples of pixels both within and immediately adjacent to the riparian zones.

4.5 Conclusions

These riparian zones in the foothills fescue grassland of southwestern Alberta within Landsat satellite image data can be accurately located (Hypothesis #2). This conclusion is in the context of the 30 m ground resolution cells resampled to 25 m.

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Chapter 5 Image Characterization Along Transects

5.1 Introduction

A monitoring tool and appropriate application techniques for assessing vegetation distribution in riparian zones across the landscape after fire is essential. Space-borne remote sensing, in the form of Landsat 7 image data, has the potential to be an effective monitoring tool given the availability of extensive coverage and multi-temporal data.

The individual, original Landsat 7 satellite image bands may not provide adequate tone differences for discrimination of land cover features across a scene (Senseman *et al.* 1996, Estes *et al.* 1982, Lillesand and Kiefer 2000). A classification groups land areas into common classes based on similar spectral patterns using two or more bands and is often applied to more efficiently differentiate similar land cover features across the scene. These spectral classes may be determined in two ways, either through a supervised or an unsupervised classification. In a supervised classification, pixels representing a known land cover feature are manually selected to create a signature file for classification based on information known about the study area. In an unsupervised classification, an algorithm is applied to create the classification with minimal user input (Lillesand and Kiefer 2000). Isoclustering (ISO) and principal component analysis (PCA), are two automated techniques to derive statistical spectral patterns within an image dataset. ISO, Iterative Self-Organizing, plants a user-defined number of centroids in the data to represent class clusters. Each pixel is then assigned to a cluster based on probability of similarity to a mean vector. The clustering algorithm runs a number of iterations until no significant change occurs in the location of mean vectors, resulting in a set of revised means. The final number of classes present in the statistics file is based on detection of

different classes, up to a maximum of the user-define number. PCA enhances the data by reducing the dimensionality within the original bands to create a set of transformed bands with less redundant information (Estes *et al.* 1982, Lillesand and Kiefer 2000). Then the classes are identified on a pixel-by-pixel basis using a maximum likelihood classifier. However, the classes may or may not match the classes identified in the field (Lillesand and Kiefer 2000, Price 1994), thereby requiring validation.

Another technique to characterize a Landsat scene is the normalized difference vegetation index (NDVI) where the ratio between band 4 (NIR) and band 3 (Red) is applied (see Table 2.2). This ratio is a commonly used index and should differentiate between types of vegetation based on vegetation health (Senseman *et al.* 1996). The NDVI represents the general reflection of green leaves on the ground. High reflection in band 4 by leaf mesophyll cells and the absorption in band 3 by chlorophyll pigments should allow the user to distinguish between woody and herbaceous vegetation. Leaf orientation of herbaceous vegetation is often vertical, as in grasses, allowing more solar energy reflection from the background cover. While woody vegetation is often more broad leaved, and perpendicular to solar energy reflecting more from the leaf than the background. Given that woody vegetation is often more dense and leafy, the solar reflection after interaction with the leaves should be brighter than if only herbaceous vegetation is present. Herbaceous vegetation, especially graminoid species are more likely to have a greater amount of background soil effect in the infrared reflection, which lowers the NDVI value (Hurcom and Harrison 1998, Elmore *et al.* 2000).

Once the characterization techniques have been applied to the satellite data, the relationship between the classified and field data needs to be established. Contingency

tables are the most frequently used statistic to determine the relationship between remotely sensed and field data (Congalton *et al.* 1983). A contingency table is a square array of numbers in rows and columns where the rows are the pixels assigned to classes by the computer and the columns are the reference data gathered from the field. Each pixel falls into a row and column, the more accurate the classification the more pixels fall into associated rows and columns. The contingency table therefore evaluates both omission (exclusion) and commission (inclusion) errors present in the classification. Using these error data the user can determine how well each class and the overall classification represents the actual field data

In most classifications, confusion between classes occurs when mixed pixels, those pixels containing more than one cover type are present within the ground resolution cell (Lillesand and Kiefer 2000). Mixed pixels are common in riparian zones because of the numerous and rapid transitions occurring between vegetation communities from moist lowlands to drier uplands.

5.2 Objectives

The objective of this portion of the study was to test the similarity between the field classes and observed satellite spectral classes along transects. The hypothesis that vegetation classes observed in the field can be identified in Landsat satellite image data using various analysis techniques was tested directly by comparing datasets through a modified contingency table technique, and indirectly by comparing the trends in average percent covers of ISO and PCA spectral classes and the site average NDVI values, to the average percent cover of the field classes along transects.

5.3 Methods

The field and Landsat 7 image data recorded along transects in 2001 and 2002 were used to test the relationship between the field and satellite data. Figure 5.1 illustrates the systematic process followed to isolate image data for analysis. The georeferenced satellite images were imported into the ArcInfo 8.0 (ESRI 1999a) Grid Environment for analysis by converting them into grids using the command *Imagegrid*. To isolate the study areas the riparian zone buffer polygons were imported from ArcView 3.2 (ESRI 1999b) using *Shapearc* and were then used to clip the grids using *Gridclip*. Once all the riparian zones were clipped from the grid, they were compiled into one grid for analysis using *Merge*. Individually each riparian zone did not contain a sufficient number of pixels for separate analyses and thus, needed to be analysed as a whole. The combined riparian zone grid data were characterized using the ISO, NDVI and PCA.

Slight changes in the solar angle, plant growth, or other temporal factors result in variation in the amount of solar radiance received and recorded by the satellite sensor. Therefore, each image needed to be characterized independent of all other image data with class numbers arbitrarily assigned for each image date. Differences between burned and unburned treatments will only be compared within the same image in 2001 and 2002 (Chapter 5), after which the change in that difference will be sequentially compared using all images (Chapter 6). Each characterization analysis used a different set of Landsat satellite image bands determined using an iterative process to identify the most accurate band combination. According to Casals-Carrasco *et al.* (2000) band selection should be based on low correlation and high covariance. Similarly, Estes *et al.* (1982) suggest that

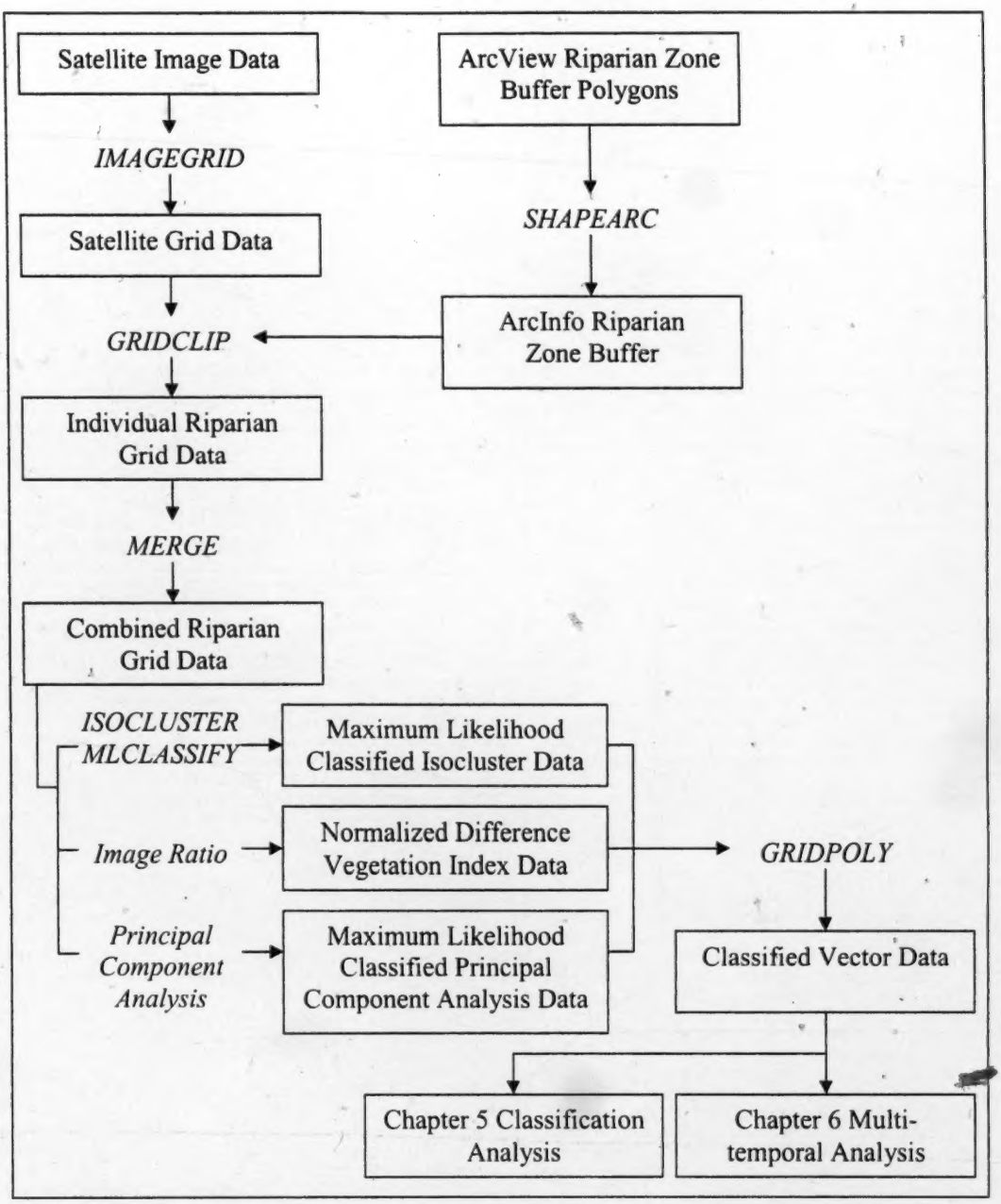


Figure 5.1 Diagram of systematic process and ArcInfo 8.0 (ESRI 1999a) commands used to classify satellite image data.

limiting an analysis to one or 2 bands results in loss of considerable information, while selecting bands that contain relevant information will alleviate this problem. In this study

the number of bands was limited to bands 3, 4 and 5 (Table 2.2). For the ISO characterization only bands 3 and 4 were used in the analysis because band 5 absorbs water and introduced considerable variability (Lillesand and Kiefer 2000). For the NDVI analysis bands 3 and 4 were used, while the PCA used bands 3, 4 and 5. Band 3, the visible red band, reflects stronger from soil than vegetation. Thus soils often are associated with lighter tones and higher digital numbers resulting in lower NDVI values. Band 4, the near-infrared band, reflects strongly from live vegetation, thereby resulting in a higher digital number when more green vegetation is present and a higher NDVI value. Thus, bands 3 and 4 should have have high covariance and low correlation for the field cover classes present.

Isoclustering was completed in the Grid Environment of ArcInfo 8.0 (ESRI 1999a). This was an appropriate technique as the spectral data in each band approximated a normal distribution. Cluster statistics for the riparian zone grid data were automatically calculated using the *Isocluster* command. The resulting cluster statistics file was then used in the *Mlclassify* command to define how each pixel should be classified based on pixel proximity to class means.

The NDVI was also calculated in the ArcInfo 8.0 Gird Environment (ESRI 1999a) using bands 3 and 4 (Equation 5.1). The NDVI is a form of continuous data and was used to compare differences between the burned and unburned treatments.

Equation 5.1. $$NDVI = \left(\left(\frac{\text{float(band 4 - band 3)}}{\text{float(band 4 + band 3)}} \right) + 1 \right) * 100$$

The PCA was completed (MathWorks, Inc. 2001) using script written by Chang. The PCA script transformed the satellite image bands 3, 4 and 5 into linear combinations from which three principal components were derived for input into the classification program. Next, the script created data clusters derived from the three principal components using a C-means and standard deviation. The script was programmed to create ten classes to reflect the number of field classes (see Table 3.2) with a maximum of 500 iterations or precision of 6 decimal places.

The field transects were identified in the riparian zones in the satellite image data (Figure 5.2). The characterized grid was converted to a polygon coverage using the command *Gridpoly* and site attributes were attached. Next, the field transect buffers (Figure 3.3) were imported using *Shapearc* and isolated using the *Intersect* command to preserve all table attributes. These data are discrete, without normal distributions, and thus were analyzed as categories using non-parametric statistics.

Contingency tables are a common technique used to assess correctness of a classification. Modified contingency tables were created with known field classes as columns and detected spectral classes as rows. The percent distribution of field classes within each spectral class was determined to associate field classes to spectral classes. When classification is 100% correct, all non-major diagonal elements should be zero. Tables were created to compare field data to ISO, NDVI and PCA characterizations along transects.

Trends of the ISO and PCA satellite image data spectral classes along transects from 2001 and 2002 were compared to the field data classes from those same years using the Friedman's Non-parametric ANOVA. In this analysis the differences in percent cover

of each class along the transects between burned and unburned treatments were compared. NDVI data along transects for 2001 and 2002 were displayed in frequency histograms and the treatment difference compared using the Kolmogrov-Smirnov Non-Parametric Test for Similarity. This test compares the similarity between the frequency histograms of each treatment.

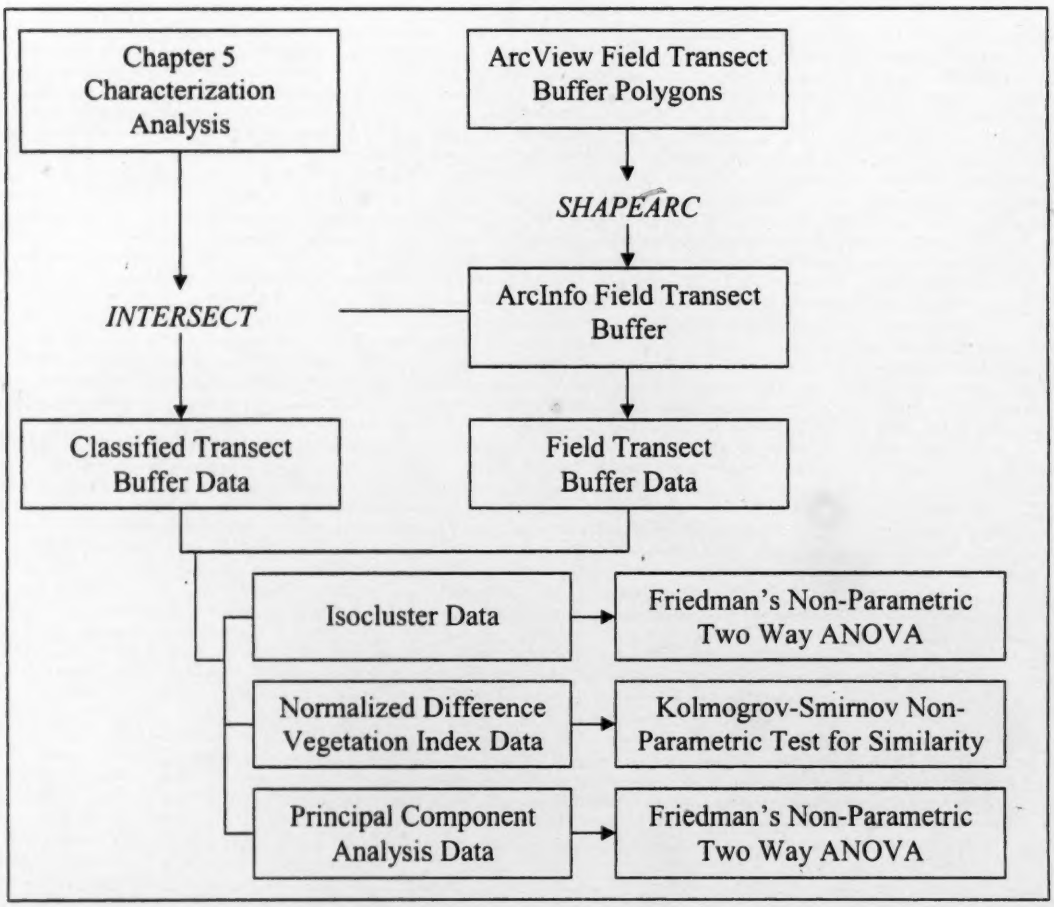


Figure 5.2 Diagram of systematic process of ArcInfo 8.0 (ESRI 1999a) commands and SAS (SAS Institute, Inc. 1999) steps used in process to statistically analyse field and characterized satellite data along transects.

5.4 Results

5.4.1 Isocluster Classification Along Transects

Using a unique set of means and covariance values (Appendix C), ISO was applied to the riparian zone dataset of each year of Landsat 7 data independently. The band 3 to band 4 scattergraphs displayed in Figure 5.3 show the location of each class centroid. The 1997 digital values were lower than the other satellite image dates, with seven classes. The 1998 image, collected in late May, had greater values for band 3 than 2001 or 2002, while 2001 and 2002 had greater band 4 values. Eight classes were identified in 1998 and 2001, while only 6 classes were identified in 2002.

The maximum likelihood classifier in ArcInfo (ESRI 1999a) also has an algorithm that estimates the number of cells correctly classified based on similarity of a pixel to its class (Table 5.1). All ISO's produced were required to have 50% and greater of the percent of pixels estimated correctly for a more accurate classification. The percent of cells estimated to be correctly classified is approximately the same for all years.

In 2001 (Table 5.2) the observed relationship of the ISO spectral classes and the field classes is not direct. Field classes were defined and identified based on a certain set of criteria (Table 3.3) that may turn out to be unrelated to very specific spectral characteristics. The variability within each spectral class in terms of different field classes being represented is greater in 2001 than 2002 (Tables 5.2 and 5.3). Spectral classes 1, 2, 6, and 8 in 2001 (Table 5.2) contain multiple dominant field classes from bare soil to lush grass and tall shrubs. Spectral class 3 is short shrub-grass dominant, classes 4 and 5 are lush grass and short shrub-grass dominant. Class 7 includes lush grass, short shrub-grass

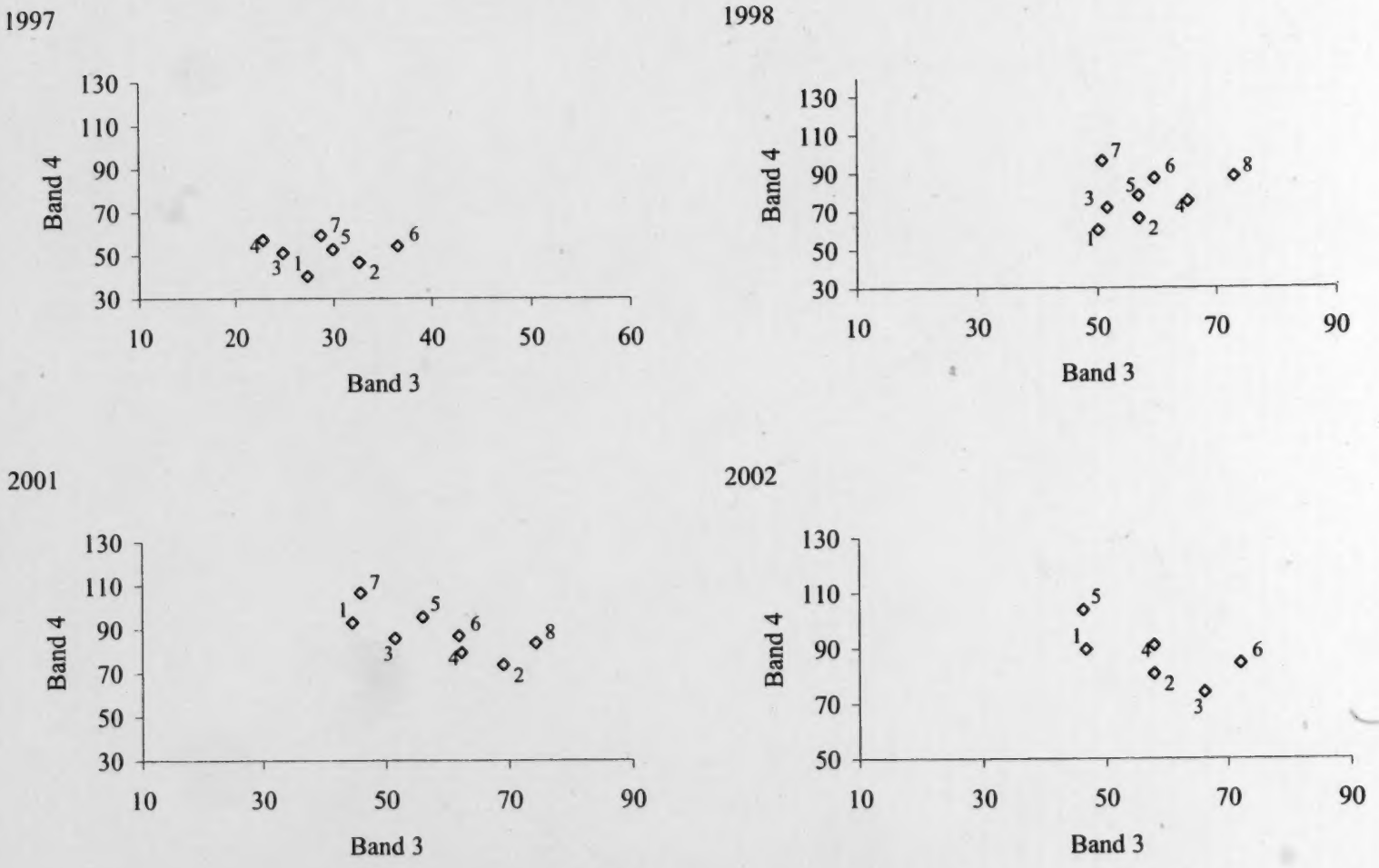


Figure 5.3 Scattergraphs of bands 3 and 4 displaying separation between ISO class centroids.

Table 5.1 Percent of cells estimated to be correctly classified in the ISO for all satellite image dates.

1997		1998		2001		2002	
Probability Correct	Percent Pixels	Probability Correct	Percent Pixels	Probability Correct	Percent Pixels	Probability Correct	Percent Pixels
1.000	0.0	1.000	0.5	1.000	0.2	1.000	0.4
0.995	0.7	0.995	0.2	0.995	0.5	0.995	0.3
0.990	1.3	0.990	0.6	0.990	0.7	0.990	0.5
0.975	1.0	0.975	1.4	0.975	0.9	0.975	2.4
0.950	1.7	0.950	3.4	0.950	3.3	0.950	2.8
0.900	10.4	0.900	11.2	0.900	9.3	0.900	11.8
0.750	18.9	0.750	22.5	0.750	22.4	0.750	23.1
0.500	30.0	0.500	29.3	0.500	31.5	0.500	29.7
0.250	19.0	0.250	18.9	0.250	17.1	0.250	15.6
0.100	4.5	0.100	3.9	0.100	4.5	0.100	3.8
0.050	3.2	0.050	1.9	0.050	2.4	0.050	2.2
0.025	2.0	0.025	2.2	0.025	1.6	0.025	1.7
0.001	1.0	0.001	1.2	0.001	1.2	0.001	0.7
0.000	6.5	0.000	2.9	0.000	4.4	0.000	5.0

Table 5.2 Modified contingency table between 2001 ISO spectral data and 2001 field data along transects, based on percent of pixels in each spectral class, where bolded numbers illustrate field class distribution within each spectral class.

Spectral Class (# samples)	Field Class									
	BSS	SGS	MGS	LGS	SSGS	MSGS	TSGS	SSS	MSS	TSS
1 (59)	**14**	0	**14**	5	**15**	**14**	5	7	**20**	7
2 (60)	**30**	3	7	**27**	7	7	0	**20**	0	0
3 (157)	3	8	6	3	**57**	5	5	0	5	8
4 (95)	6	6	**29**	**21**	**26**	1	2	1	3	3
5 (183)	4	3	**10**	**20**	**31**	0	7	**15**	**5**	**4**
6 (175)	**15**	**12**	**14**	**12**	**26**	1	0	**10**	**5**	**5**
7 (74)	0	0	0	**45**	**15**	3	**27**	9	1	0
8 (39)	3	**10**	**10**	0	**31**	**21**	**21**	3	0	3

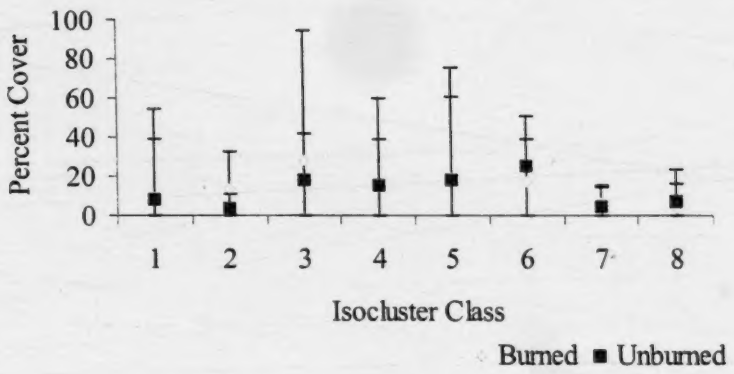
and tall shrubs. In 2002 (Table 5.3), most of the spectral classes were dominated by moderate to lush grass and short shrubs. Classes 1 and 4 also hae taller shrubs present. In contrast, class 6 is bare soil and moderate grass cover. Direct relationships between the ISO spectral classes and field classes were not observed.

Table 5.3 Modified contingency table between 2002 ISO spectral data and 2002 field data along transect, based on percent of pixels in each field class, where bolded numbers illustrate field class distribution within each spectral class.

Spectral Class (# samples)	Field Classes									
	BSS	SGS	MGS	LGS	SSGS	MSGS	TSGS	SSS	MSS	TSS
1 (507)	2	0	**26**	**17**	**20**	6	0	**11**	2	**15**
2 (823)	5	0	**37**	**24**	**17**	3	1	7	2	3
3 (390)	4	2	**59**	**12**	**14**	1	0	5	1	2
4 (721)	7	0	**32**	**23**	**21**	1	2	**10**	**1**	**3**
5 (762)	4	0	**15**	**41**	**18**	3	0	8	6	3
6 (221)	**22**	0	**49**	10	9	1	0	5	0	4

Applying Friedman's Non-Parametric ANOVA to the ISO transect data resulted in no significant difference in treatment effects in 2001 (Figure 5.4). Comparing the 2001 ISO spectral data (Figure 5.4) to the 2001 field data (Figure 3.6), neither have a significant difference in treatment effect. The ISO classes indicate that the cover of short woody shrubs is greater in the burned treatment, agreeing with field observations. In 2002 (Figure 5.4), ISO class 2 was significantly greaterer and class 4 was significantly lower in the burned area. The ISO classes show that greater percent cover of moderate to lush grass and short shrubs were present in the burned area in 2002, which reflects the field observations (Figure 3.6). Both the ISO and field data for 2002 indicate less cover of tall shrubs in the burned area.

2001



2002

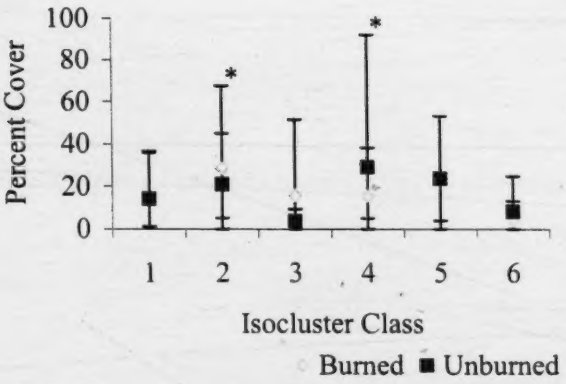


Figure 5.4 Mean percent cover of each ISO class along transects for burned and unburned treatments, where the y-error bars show the minimum and maximum range of percent cover within each ISO class.

5.4.2 Normalized Difference Vegetation Index Along Transects

NDVI characterizes the data, based on greenness of the land cover, over a continuous gradient, not discrete classes as in the ISO and PCA. Classes are created after the NDVI values are assigned based on groupings of field classes (see Table 3.2) along a continuous range of values. However, the contingency tables for 2001 (Table 5.4) and

2002 (Table 5.5) showed little direct relationship between field classes and spectral classes. In 2001 (Table 5.4) there was a large variation of field classes along the NDVI data (grouped into classes of five values wide for discussion purposes only). The range of values for moderate to lush grass, and short shrub-grass occur over large ranges of NDVI values, while the shrub communities occur intermittently. Bare soil overlaps with the grass and shrub-grass cover at lower NDVI values, while shrub communities were more frequent at higher NDVI values. This broken distribution prevents the division of NDVI values into discrete classes representing field classes.

The NDVI values for 2002 (Table 5.5) are less variable than 2001 (Table 5.4), but the distribution still did not provide a basis for creating discrete classes. The lower NDVI values represent bare soil, then increase to represent moderate to lush grass and short shrub-grass communities, while the greater NDVI values represent shrub communities.

Table 5.4 Modified contingency table between 2001 NDVI spectral data and 2001 field data along transects, based on percent of pixels in each spectral ratio group, where bolded numbers illustrate field class distribution within each grouping of NDVI values.

Spectral Ratio (# samples)	Field Classes									
	BSS	SGS	MGS	LGS	SSGS	MSGS	TSGS	SSS	MSS	TSS
95-99 (5)	0	0	0	0	0	**100**	0	0	0	0
100-104 (28)	**11**	**29**	0	**50**	**11**	0	0	0	0	0
105-109 (94)	**26**	1	**12**	**11**	**22**	**9**	**11**	**10**	0	0
110-114 (125)	4	**11**	**24**	**16**	**30**	7	2	1	2	4
115-119 (180)	**19**	**7**	**17**	**12**	**15**	**7**	1	**10**	1	11
120-124 (215)	**7**	**6**	**13**	**10**	**38**	0	0	0	0	1
125-129 (141)	3	0	**8**	**18**	**44**	0	4	**13**	**7**	**4**
130-134 (96)	5	0	**11**	**6**	**36**	2	**8**	**5**	**8**	**17**
135-139 (54)	4	0	4	**46**	7	7	2	**20**	**9**	0
140-144 (41)	0	0	0	**44**	0	7	**46**	0	2	0
145-149 (1)	0	0	0	0	0	0	0	0	**100**	0

Table 5.5 Modified contingency table between 2002 NDVI spectral data and 2002 field data along transects, based on percent of pixels in spectral ratio group, where bolded numbers illustrate field class distribution within each grouping of NDVI values.

Spectral Ratio (# samples)	Field Classes									
	BSS	SGS	MGS	LGS	SSGS	MSGS	TSGS	SSS	MSS	TSS
80-84 (2)	**100**	0	0	0	0	0	0	0	0	0
85-89 (6)	**83**	0	17	0	0	0	0	0	0	0
90-94 (26)	**46**	0	**46**	4	4	0	0	0	0	0
95-99 (95)	**22**	0	**47**	3	**18**	0	0	8	0	1
100-104 (127)	8	2	**53**	**20**	6	1	0	4	2	4
105-109 (236)	**11**	2	**51**	9	**16**	2	0	6	0	3
110-114 (507)	5	0	**44**	**22**	**15**	1	1	6	2	4
115-119 (477)	5	2	**34**	**29**	**16**	1	4	5	1	2
120-124 (689)	7	1	**34**	**20**	**17**	2	1	**11**	1	7
125-129 (468)	7	0	**24**	**19**	**21**	2	0	**16**	3	7
130-134 (445)	6	0	**29**	**25**	**17**	5	0	**10**	1	6
135-139 (278)	1	3	**19**	**30**	**19**	3	0	**13**	**9**	1
140-144 (226)	3	0	**10**	**46**	**23**	0	1	**8**	**8**	0
145-149 (73)	0	0	1	**67**	8	0	0	**11**	**3**	**10**
150-154 (68)	0	0	4	**29**	**44**	0	0	1	0	**21**

Applying a Kolmogrov-Smirnov Non-Parametric Test for Similarity to the NDVI data resulted in a significant difference between burned and unburned areas in sites 3 and 5 in 2001 and sites 1, 5 and 6 in 2002 (Figure 5.7). The frequency histograms for 2001 are displayed in Figure 5.5 and for 2002 in Figure 5.6 showing the range of NDVI values at each site. The NDVI value was significantly higher at site 3 and lower at site 5 in the burned area in 2001 (Figures 5.5 and 5.7). In 2002, the NDVI value was significantly higher at site 1, lower at site 5 and lower at site 6 in the burned area (Figures 5.5 and 5.7). The NDVI values in the burned treatment increased at 4 sites and decreased at 3. The increase in NDVI value suggests a trend to greater bare soil or herbaceous vegetation

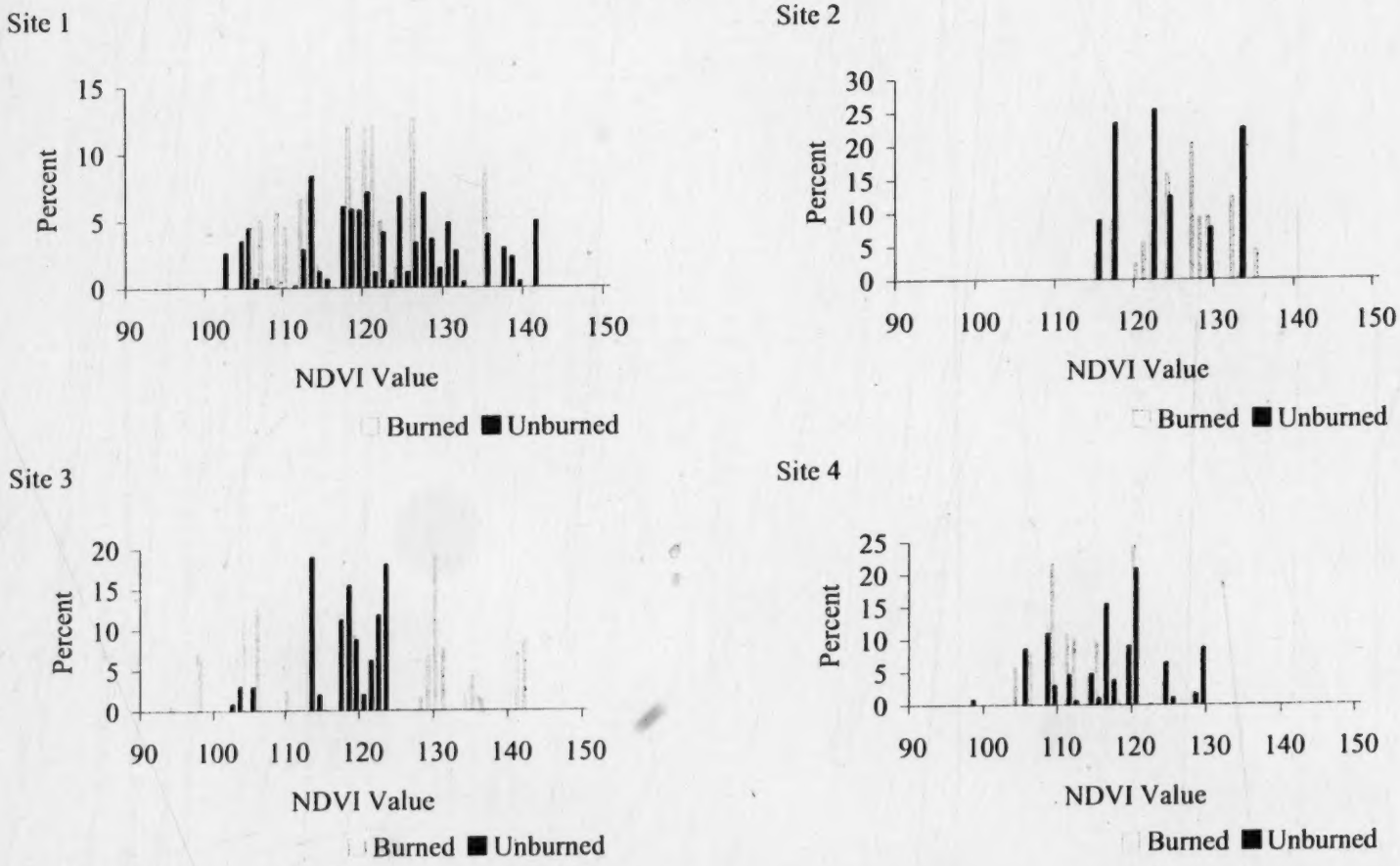
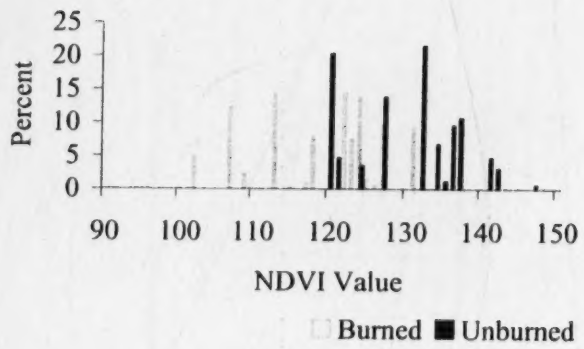
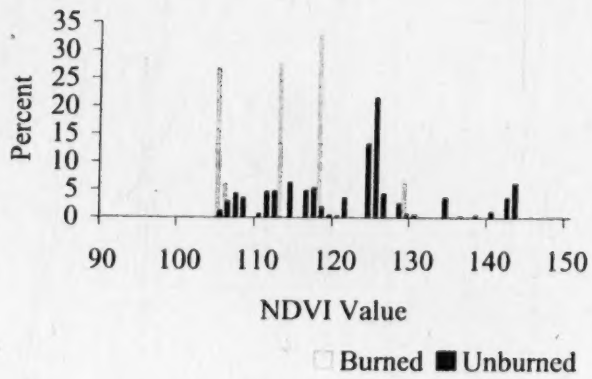


Figure 5.5 Distribution of NDVI values along transects for sites 1 to 4 in 2001 according to percent of pixels at each value.

Site 5



Site 6



Site 7

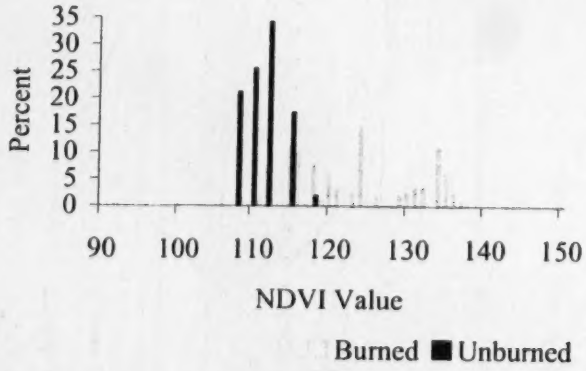


Figure 5.5 (continued) Distribution of NDVI values along transects for sites 5 to 7 in 2001 according to percent of pixels at each value.

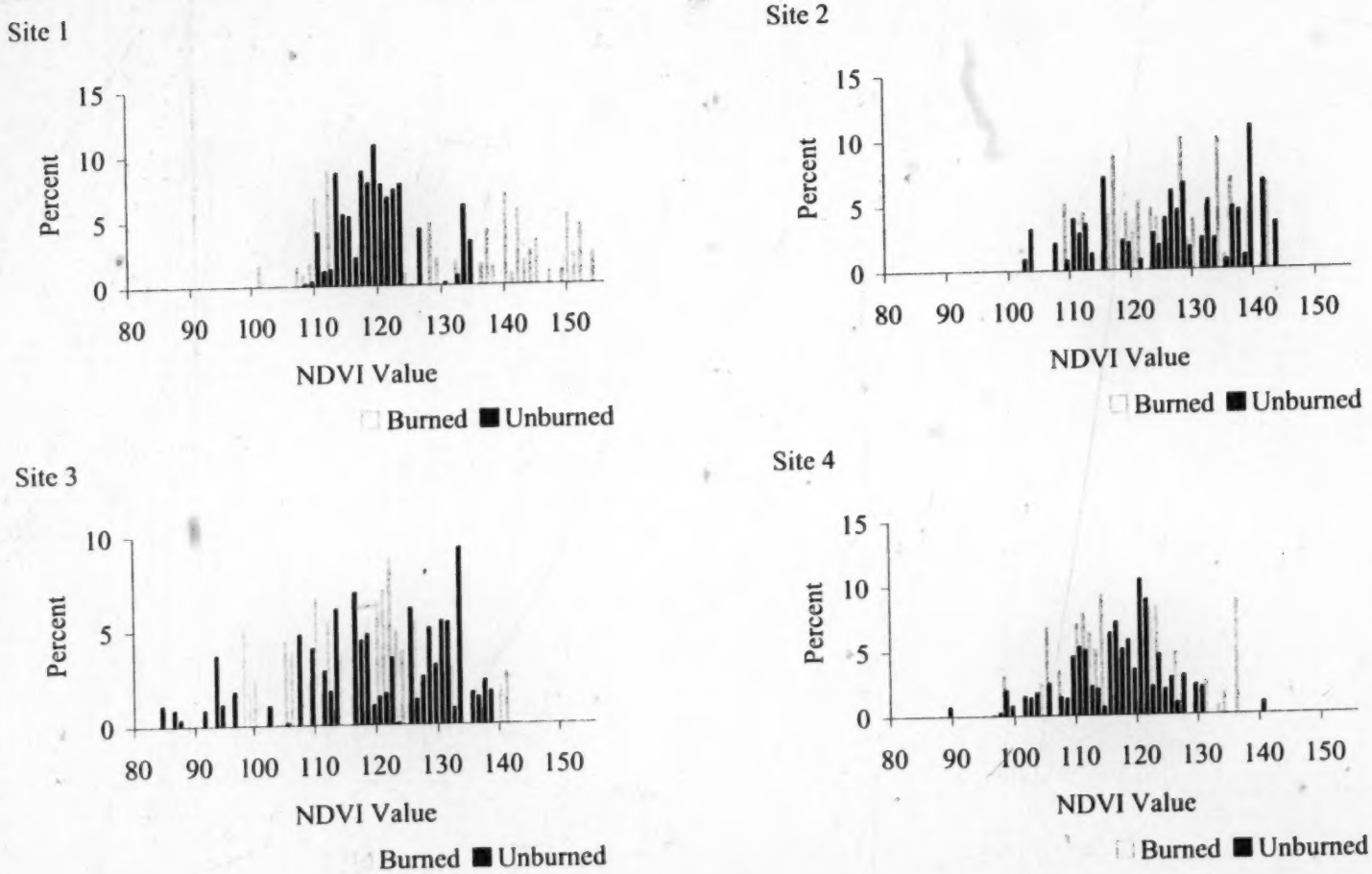
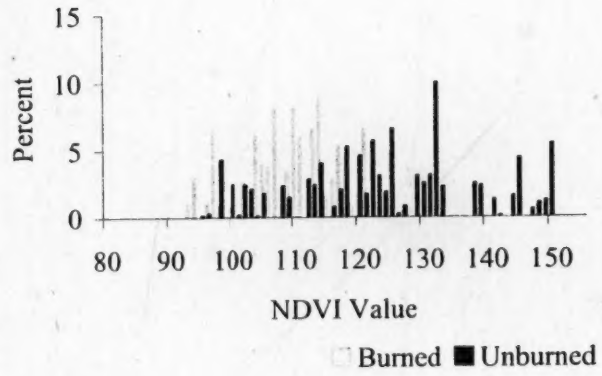
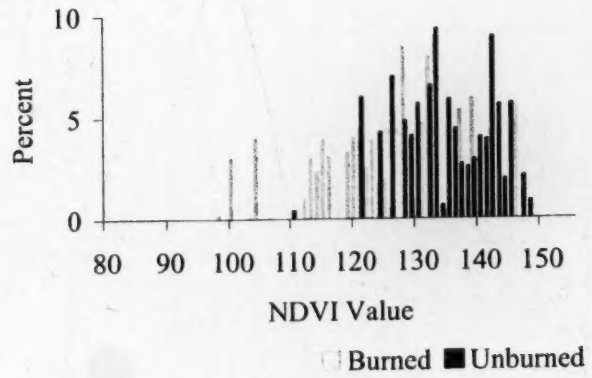


Figure 5.6 Distribution of NDVI values along transects for sites 1 to 4 in 2002 according to percent of pixels at each value.

Site 5



Site 6



Site 7

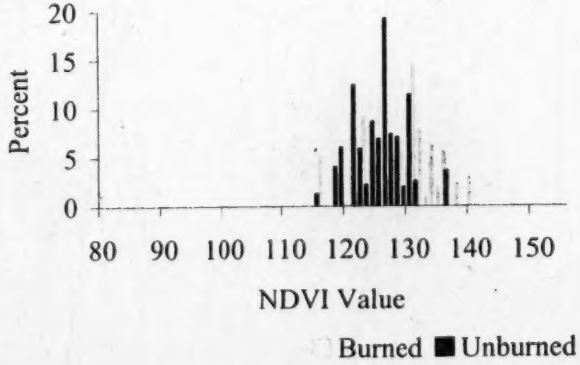


Figure 5.6 (continued) Distribution of NDVI values along transects for sites 5 to 7 in 2002 according to percent of pixels at each value.

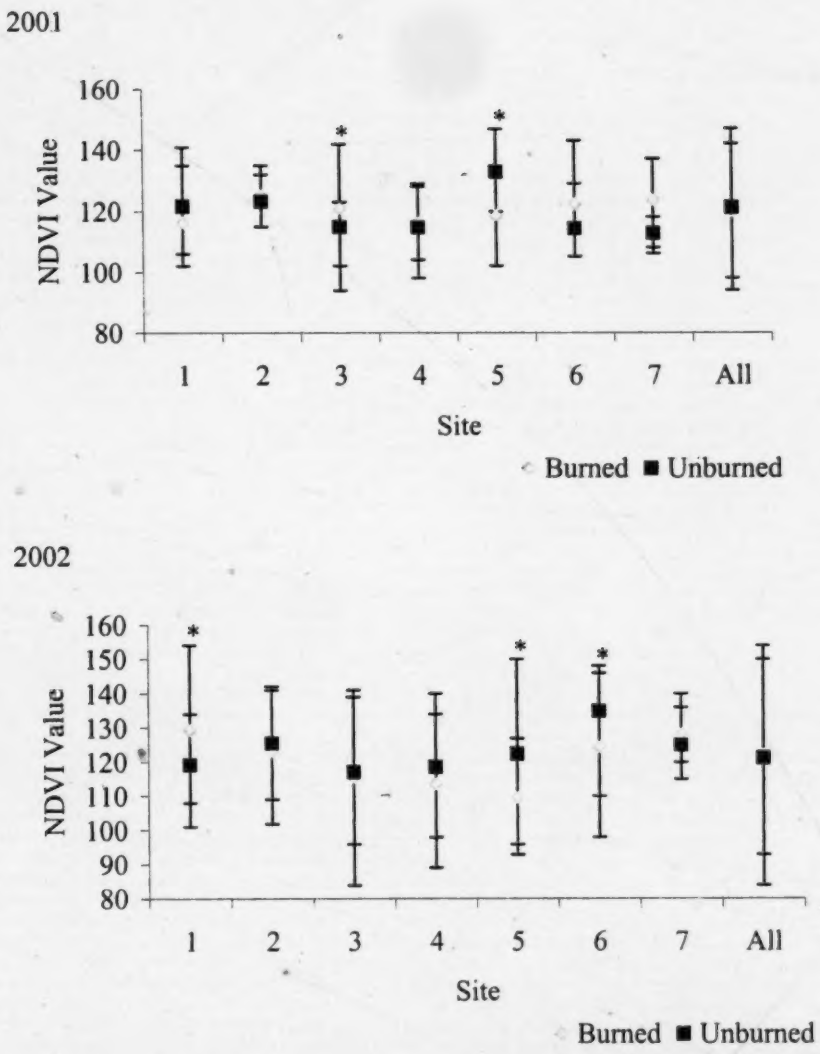


Figure 5.7 Mean NDVI value along transects for burned and unburned treatments, where the y-error bars show the minimum and maximum range of percent cover within each field class, and the * indicates significant treatment effect between distribution histograms.

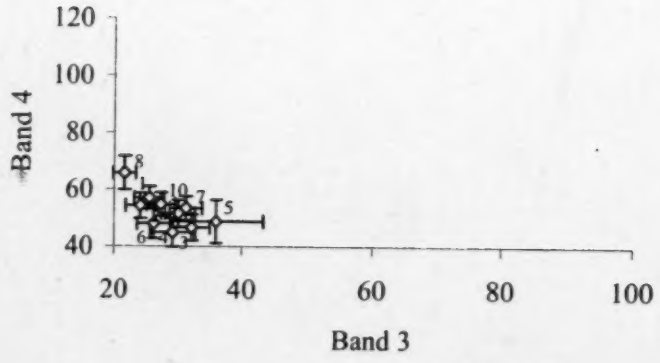
cover, while a decrease suggests a trend to greater woody vegetation cover. From 2001 to 2002, the moderate to lush grass, shrub-grass and shrub communities increased at site 1, decreased at sites 3, 4, 6 and 7, and did not change at sites 2 and 5. The overall NDVI

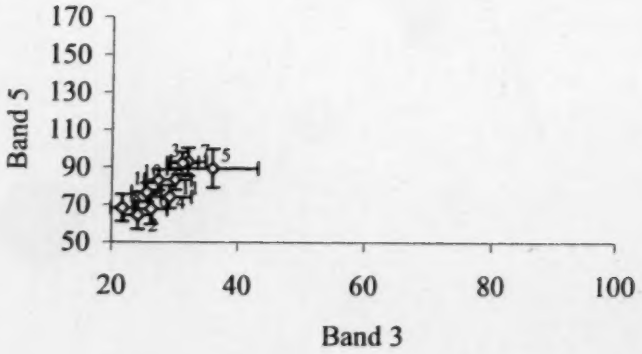
values indicated no difference between the burned and unburned areas in 2001 or 2002 while the field data show an increase in herbaceous and woody vegetation cover in the burned area.

5.4.3 Principal Component Analysis Along Transects

PCA was applied to each year of data independently using a unique set of means, standard deviations and covariance values derived for that year of data. The entire study area was used in the PCA to keep a consistent classification across the entire study area in addition to along the transects. Because three satellite image bands were used in the PCA, each year required three scattergraphs to clearly display class separability. In 1997 (Figure 5.8), PCA classes were tightly clustered and at lower digital values, while in 1998 (Figure 5.9), 2001 (Figure 5.10) and 2002 (Figure 5.11) the spread of class clusters were more open and at higher digital values. Considerable overlap between classes within one standard deviation was observed with the two-band scattergraphs for all years, however, with three band combinations the data were more separable. Note that PCA actually derives 11 spectral classes, but one of class number 9, 10 or 11 is not displayed in the dataset because it represents nodata and thus was not included in the analysis.

Similar to the modified contingency tables for NDVI and ISO, neither the 2001 (Table 5.6) or 2002 (Table 5.7) PCA spectral classes were directly related to one field class (see Table 3.2), but instead represented multiple vegetation classes. In 2001, many field classes were mixing in PCA spectral classes (Table 5.6). Class 8 was specifically moderate height shrub-grass vegetation community, but contained only 3 samples.





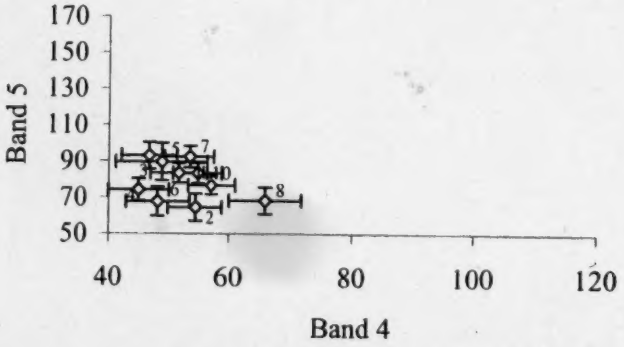
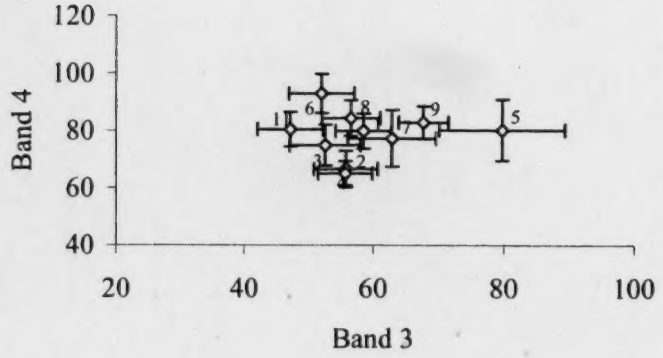
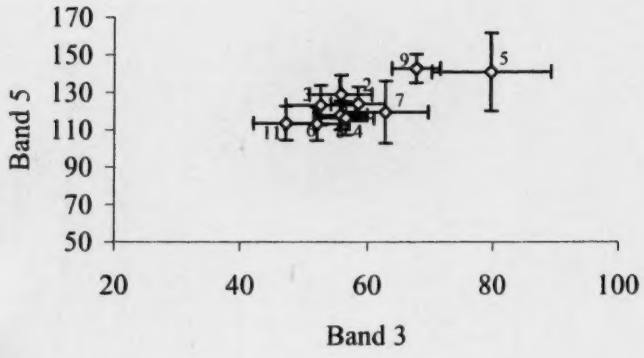


Figure 5.8 Scattergraphs of combinations of bands 3, 4 and 5 from 1997 displaying separation between classes derived from the PCA where the x and y-error bars represent one standard deviation.





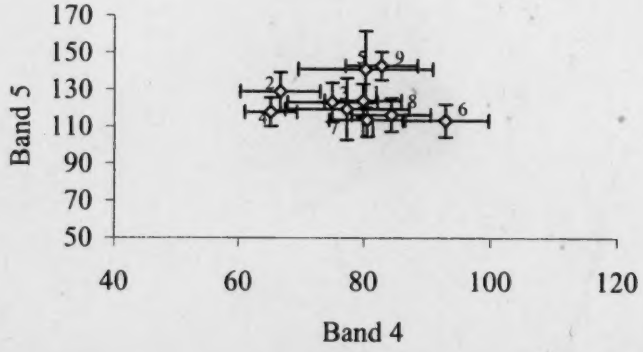


Figure 5.9 Scattergraphs of combinations of bands 3, 4 and 5 from 1998 displaying separation between classes derived from the PCA where the x and y-error bars represent one standard deviation.

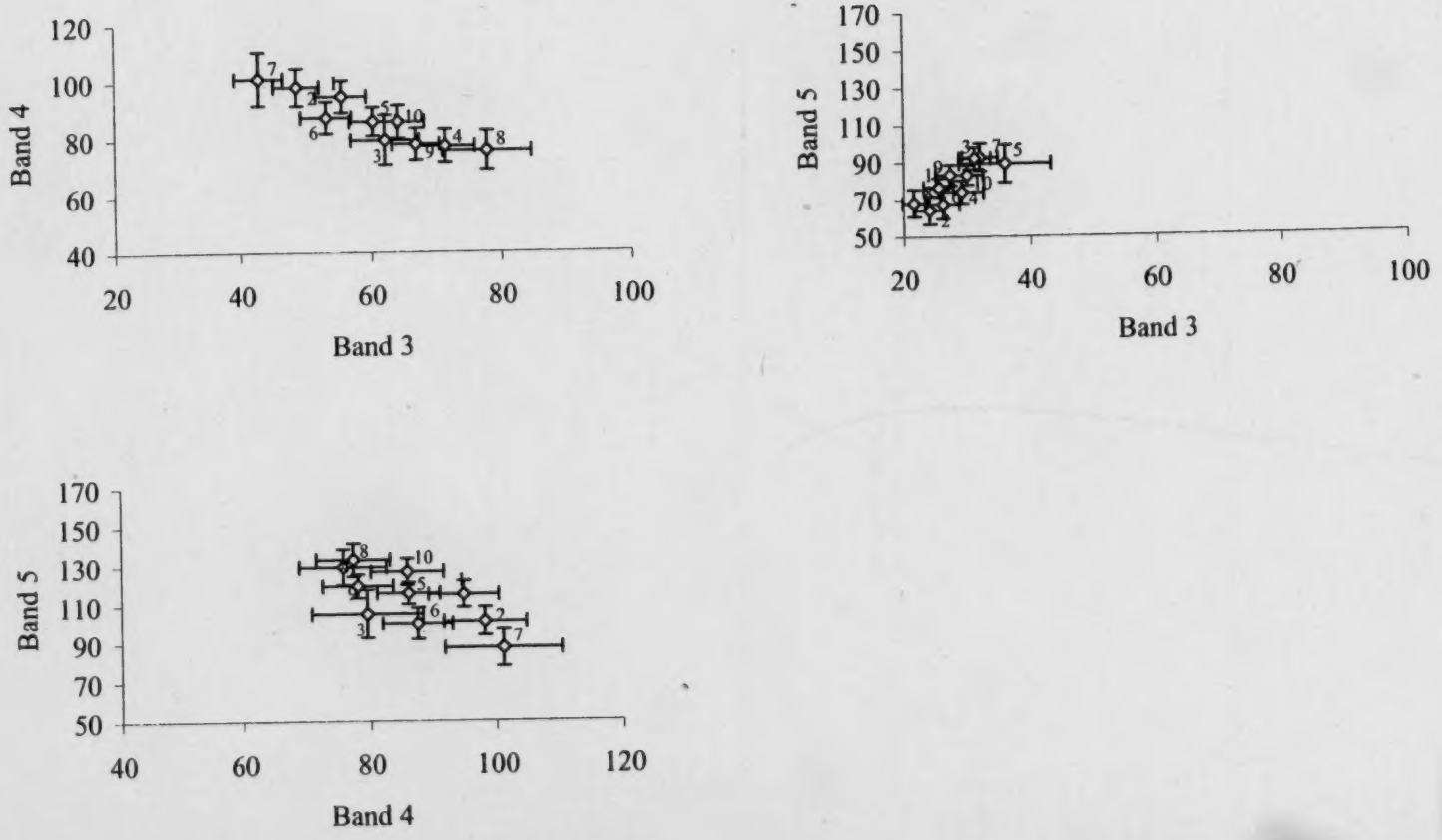


Figure 5.10 Scattergraphs of combinations of bands 3, 4 and 5 from 2001 displaying separation between classes derived from the PCA where the x and y-error bars represent one standard deviation.

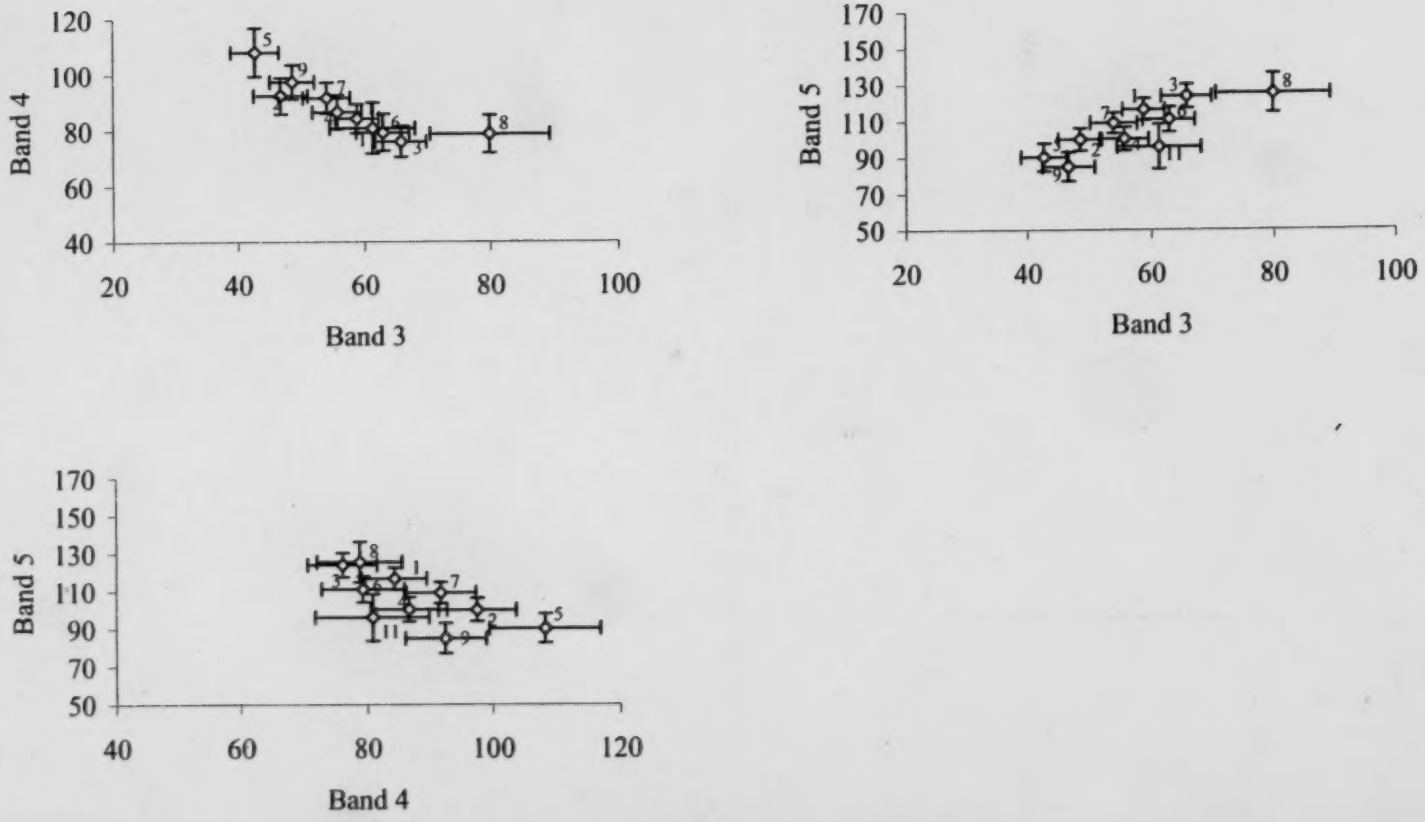


Figure 5.11 Scattergraphs of combinations of bands 3, 4 and 5 from 2002 displaying separation between classes derived from the PCA where the x and y-error bars represent one standard deviation.

Classes 3, 4, 5, 9 and 10 were bare soil, grass and short shrub-grass, while classes 1, 2, 6, and 7 were mainly lush grass, short shrub-grass, and shrubs. In 2002 (Table 5.7), spectral classes 1, 3, 4 and 6 were all moderate to lush grass and short shrub-grass While classes 2, 5, 7, and 9 were moderate to lush grass and short shrub-grass as well as shrubs. Classes 8 and 11 were mainly moderate to lush grass but also contained bare soil and some shrub-grass. In 2002, the field classes seem spectrally similar, as in the ISO and NDVI.

Table 5.7 Contingency table between 2001 PCA spectral class data and 2001 field data along transects, based on percent of pixels in each spectral class, where bolded numbers illustrate field class distribution within each spectral class.

Spectral Class (# samples)	Field Class									
	BSS	SGS	MGS	LGS	SSGS	MSGS	TSGS	SSS	MSS	TSS
1 (132)	3	2	**14**	**17**	**36**	0	**10**	**13**	0	6
2 (121)	7	0	8	**27**	**25**	0	**13**	**15**	2	3
3 (67)	**13**	0	**12**	**19**	**25**	7	3	**12**	4	3
4 (28)	**32**	0	**21**	**7**	**36**	0	0	4	0	0
5 (111)	**16**	3	**20**	3	**40**	4	0	7	0	8
6 (134)	3	0	**19**	**7**	**37**	7	6	3	**16**	3
7 (59)	0	0	0	**29**	**15**	**14**	**14**	2	**27**	0
8 (3)	0	0	0	0	0	**100**	0	0	0	0
9 (58)	**10**	**14**	**17**	7	**24**	**14**	0	**14**	0	0
10 (134)	**10**	**22**	**28**	**10**	**18**	0	6	1	0	7

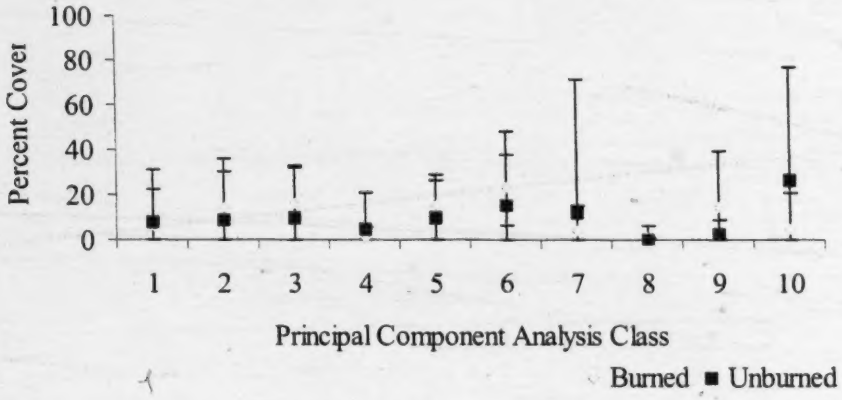
Friedman's Non-Parametric ANOVA (SAS Institute, Inc. 1999) was applied to the PCA classification results along the transects to assess significant differences in PCA classes between the burned and unburned areas. Similar to the field class statistics, there were no significant differences in spectral classes between burned and unburned areas in 2001 or 2002 (Figure 5.12). In 2001, the percent cover of woody vegetation was greater in the burned area indicated by classes 2 and 6, lower in class 7, and was not different in

Table 5.8 Contingency table between 2002 PCA spectral class data and 2002 field data along transects, based on percent of pixels in each spectral class, where bolded numbers illustrate field class distribution within each spectral class.

Spectral Class (# samples)	Field Class									
	BSS	SGS	MGS	LGS	SSGS	MSGS	TSGS	SSS	MSS	TSS
1 (341)	3	1	**44**	**23**	**22**	0	0	5	0	2
2 (496)	5	0	**25**	**30**	**23**	0	0	**11**	**3**	**3**
3 (213)	2	2	**43**	**30**	**17**	0	0	1	0	5
4 (375)	8	0	**29**	**23**	**21**	1	0	7	3	9
5 (369)	2	0	**13**	**43**	**22**	0	1	**7**	**6**	**7**
6 (323)	5	1	**52**	**15**	**15**	2	0	7	1	3
7 (486)	3	0	**33**	**25**	**24**	0	0	**15**	0	0
8 (181)	**23**	0	**55**	4	9	0	0	7	0	2
9 (272)	6	0	**15**	**15**	**10**	**13**	1	**17**	**6**	**18**
11 (338)	**11**	0	**38**	**17**	9	7	7	5	2	4

class 1. The other cover and herbaceous vegetation was higher in the burned area indicated by classes 3, 5 and 9, and lower in classes 4 and 10. In 2002, classes 1 through 5, 7 and 9 showed little difference in percent cover of woody vegetation between the burned and unburned areas, while class 6 indicated an increase in lush grass and shrub cover in the burned area. Class 8 and 10 showed little difference in percent cover of herbaceous vegetation and other cover between burned and unburned areas. In 2001 and 2002, field data (Figure 3.6) indicate an increase in percent cover of herbaceous and woody vegetation in the burned area which is reflected in the PCA classes by higher proportion of spectral classes representing moderate to lush grass, shrub-grass, and shrub cover in the burned area.

2001



2002

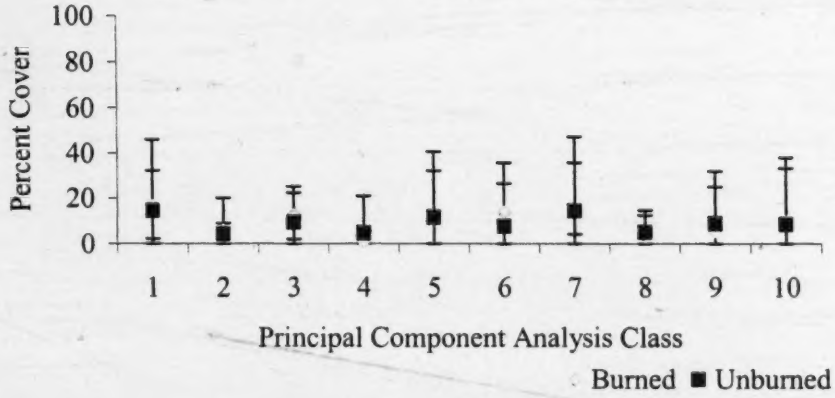


Figure 5.12 Mean percent cover of each PCA class along transects for burned and unburned treatments, where the y-error bars show the minimum and maximum range of percent cover within each class.

5.5 Discussion

5.5.1 Isocluster Classification

When comparing the ISO spectral classes to the field classes along transects, the contingency table showed no direct relationship. Casals-Carasco *et al.* (2000) state that further analyses are not applicable if the contingency table does not represent the Landsat

7 scene. However, the nature of this study with multi-temporal satellite images allows analyses of changes between image years.

Although significant differences in percent cover along the transects were observed in the 2002 ISO derived spectral classes, these differences were very small and agreed with the trend of increasing woody vegetation observed in field data. The decreased number of classes in 2002 compared to 2001 suggests that the field classes selected were more spectrally similar (Lillesand and Kiefer 2000, Price 1994). The significant differences observed in 2002 may be a result of there being fewer spectral classes than field classes. Field classes were selected based on fine-scale differences observed, while Landsat 7 spectral class are coarser (Lillesand and Kiefer 2000).

Trends in data between the field and ISO classes along the transects from 2001 to 2002 indicated the percent cover of moderate to lush grass, shrub-grass and shrub cover increased in the burned area. The ISO classes reflected the increase in those field classes. The similarity of spectral classes occurs when field classes were defined and identified on a certain set of criteria that turn out to be unrelated to the very specific spectral characteristics. Therefore the spectral and field classes are not generally comparable, which is a common outcome of an unsupervised classification. The detection of separate spectral classes indicates that there are differences on the ground but which relate to some factor other than the field classes investigated. The greater variability in the 2001 ISO classes may have been a function of the drought conditions, while in 2002 the more similar spectral classes were a function of greater precipitation increasing vegetation growth and its homogeneity of distribution. The presence of mixed field classes within the pixel also is certain to cause confusion, increasing the number of field vegetation

classes represented in a spectral class (Lillesand and Kiefer 2000, Smith *et al.* 2002, Townshend 2000). According to Smith *et al.* (2002), classification accuracy decreases as patch size decreases and as landscape heterogeneity increases. Riparian zones were very heterogeneous with respect to vegetation composition and topography resulting in a very patchy landscape with numerous edges. Every edge classified in a satellite image is a mixed pixel (Townshend 2000), and any patch less than 10 pixels in size has only a 15-25% chance of being correctly classified (Smith *et al.* 2002).

5.5.2 Normalized Difference Vegetation Index

Similar to the ISO analysis, the NDVI data did not relate well with the field classes suggesting that the chosen field classes were not spectrally discernable when using NDVI. NDVI differentiates well between brightness values across a Landsat scene, but features with similar spectral patterns at different intensities will have similar NDVI values because the ratio of band 4 to 3 will be similar (Lillesand and Kiefer 2000). All field transects across the riparian zones were observed to be variable in NDVI frequency distribution, which may explain the lack of relationship between the index data and field classes. The decreased variability observed between field classes and NDVI data in 2002 may be a function of increased moisture for greater vegetation growth.

For all transects in the experimental riparian zones, the average NDVI value was not significantly different in 2001 or 2002 suggesting that there was little difference between burned and unburned areas. However, significant differences between burned and unburned areas were observed at sites 3 and 5 in 2001 and 1, 5 and 6 in 2002. In 2001 suggesting greater variation within individual sites. In contrast to the field class

data, the NDVI data was analysed within individual sites indicating where the main differences across the study area occurred. The increase in percent cover of grass, moderate shrubs and tall shrubs in the field was reflected in the increased average NDVI value at some sites. Overall, there was no significant difference across the entire study area suggesting that the NDVI data classification did not provide an adequate representation of the field sample data. Oleson *et al.* (1995) observed that there is considerable spectral overlap between burned and unburned landscapes because of vegetation regrowth resulting in the inability to distinguish the two treatments. McGuire *et al.* (2000) found that quantifying vegetation abundance on semi-arid landscapes was difficult because the spectral reflectance of sparse vegetation cover is heavily influenced by soil interactions. Thus, the observed minimal change in NDVI across the landscape may be due to a lush vegetation cover on both treatment areas in both years and not necessarily different types of vegetation cover.

5.5.3 Principal Component Analysis

A comparison of the field classes to the PCA spectral classes along transects resulted in no clear relationship, meaning that the PCA classes could not be directly attributed to any specific field class, but rather represented multiple field classes. Neither the field classes nor the PCA classes differed significantly between the burned and unburned areas. The increased cover of herbaceous and woody vegetation was observed in both field and PCA classes, suggesting the PCA classification seems to adequately reflect the field data. In 2002, the strong relationship between the spectral classes and the moderate to lush grass and short shrub-grass classes may have been a function of the high

number of samples for those classes. Precipitation in 2002 was also greater than in 2001, which may have increased vegetation growth and homogeneity of distribution. The discussion about mixed pixels in section 5.5.1 applies here too.

Given that the PCA reduces the dimensionality of the dataset, it should enhance the resulting classification and be more accurate than the ISO (Lillesand and Kiefer 2000). However, the PCA data may not directly represent the field classes because of spectral similarity between the later. Furthermore, becasue the PCA may represent land cover other than the selected seral stage field classes, perhaps it is recording information about range productivity, biomass, or range conditions (Smith *et al.* 1994).

5.6 Conclusions

General differences observed in the field between burned and unburned areas of these selected riparian zones may be detected using ISO and PCA derived spectral classes, but not NDVI data. The derived spectral classes, while not specific to one vegetation type as defined in the field, did represent groupings of vegetation types. Specific vegetation differences measured in the field were not identified in the Landsat spectral classes (Hypothesis #3).

5.7 Literature Cited

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Chapter 6 Multi-temporal Analysis of Selected Riparian Zones

6.1 Introduction

The complex nature of the distribution of rangeland and riparian plants with respect to phenology, topography, moisture and small size within a mosaic over the landscape creates a highly variable pattern to analyze. According to Stohlgren *et al.* (1998), heterogeneity is the underlying patchiness of the landscape. This patchiness includes variation in nutrient levels, topography, moisture gradients, disturbances by ground squirrels (*Spermophilus richardsonii*), ungulates and other animals, seedbank composition and chance establishment, competition, fire and heterogeneous herbivory and temporal variation (succession, invasion and climate change). The high complexity of the natural vegetation arrangement has created many difficulties associated with separating the spectral reflectance patterns of the pixel sub-components into vegetation, edaphic, topographic and moisture features (Stohlgren *et al.* 1998, Van Kootwijk *et al.* 1995). However, the increasing refinement of satellite sensors is improving space borne measurement of vegetation spectral reflectance patterns with higher resolution and accuracy. Satellite monitoring of rangeland condition has improved dramatically over the last 30 years to account for the variability of vegetation, topography and moisture present in the rangeland. Overall, satellite image data have the potential to be an effective tool for monitoring changes in rangeland and riparian zones.

6.2 Objectives

The objective of this classification section was to assess land cover change between 1997 and 2002 using the ISO, NDVI and PCA spectral characterizations (Figure

1.2). The hypothesis that multi-temporal Landsat satellite image data can be used to detect changes in vegetation cover over time will be tested by comparing the difference in average percent cover of each spectral class and patch statistics of those classes in burned and unburned treatments over time.

6.3 Methods

Landsat satellite image data were acquired for 1997, 1998, 2001 and 2002 (Table 2.2). Riparian zones in the study area were identified (Figures 3.1 and 3.2), isolated (Figure 5.1) and characterized using ISO, NDVI and PCA. To compare the observed treatment effects as observable with the ISO and PCA classification techniques, the percent cover of each class within the treatment areas was derived to standardize the data for comparison. A Paired-Comparison T-Test was used (SAS institute, Inc. 1999) to compare the difference in percent cover of each class between burned and unburned treatments. The number of patches and average patch size were calculated to compare patch statistics between treatments. Patches were derived from the ArcInfo (ESRI 1999a) coverages and exported to a spreadsheet for analysis. A patch is a contiguous grouping of similar pixels associated in horizontal and vertical directions, while pixels aligned diagonally are separate patches. For the NDVI, the Kolmogrov-Smirnov Test for Similarity was completed (SAS Institute, Inc. 1999) to compare the frequency distribution of NDVI values between burned and unburned treatments at each site for each image year. The approach is outlined in Figure 6.1.

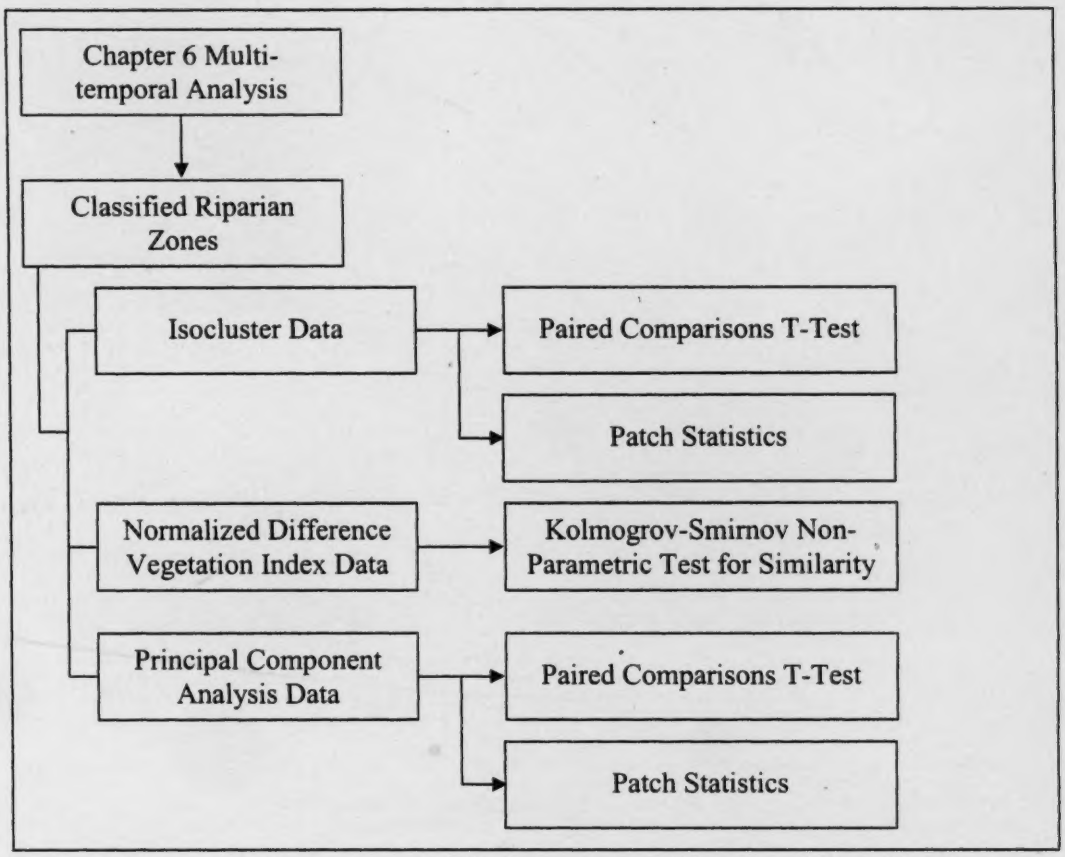

Figure 6.1 Diagram of process to statistically analyze multi-temporal classified data.

6.4 Results

6.4.1 Isocluster Classification Analysis

Figures 6.2 through 6.5 display the ISO image data classification for the riparian zones. Given that class numbers within each year were arbitrarily assigned, colours and class numbers cannot be directly compared between years.

The distribution of ISO spectral classes in 2001 and 2002 are illustrated in Figures 6.4 and 6.5 with colours defined in Table 6.3. In 2001, class 4 (blue), lush grass and

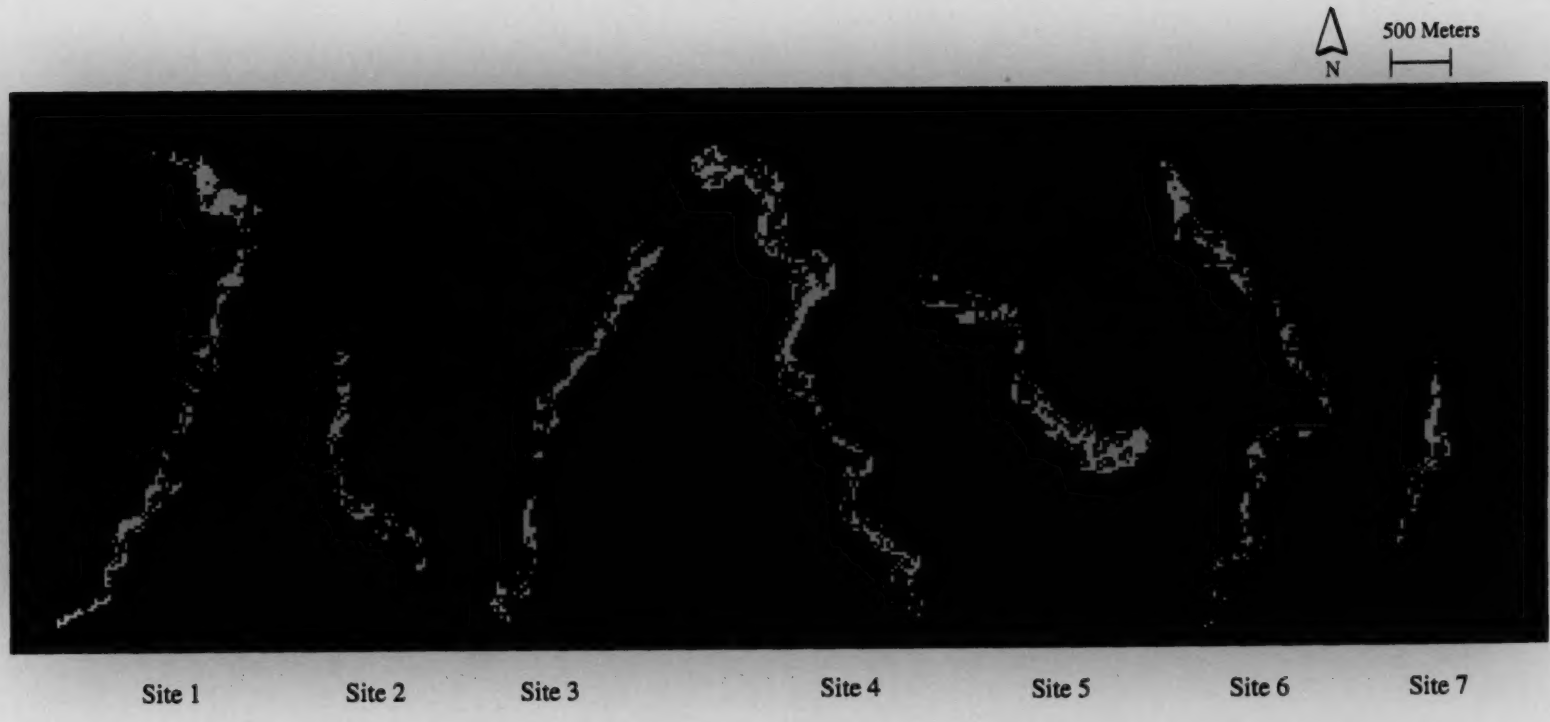

Figure 6.2 1997 ISO classification of selected riparian zone sites, where the grey line indicates the fire boundary, B represents the burned treatment and U the unburned treatment (Sites are in numeric order, not geographically oriented across the landscape).

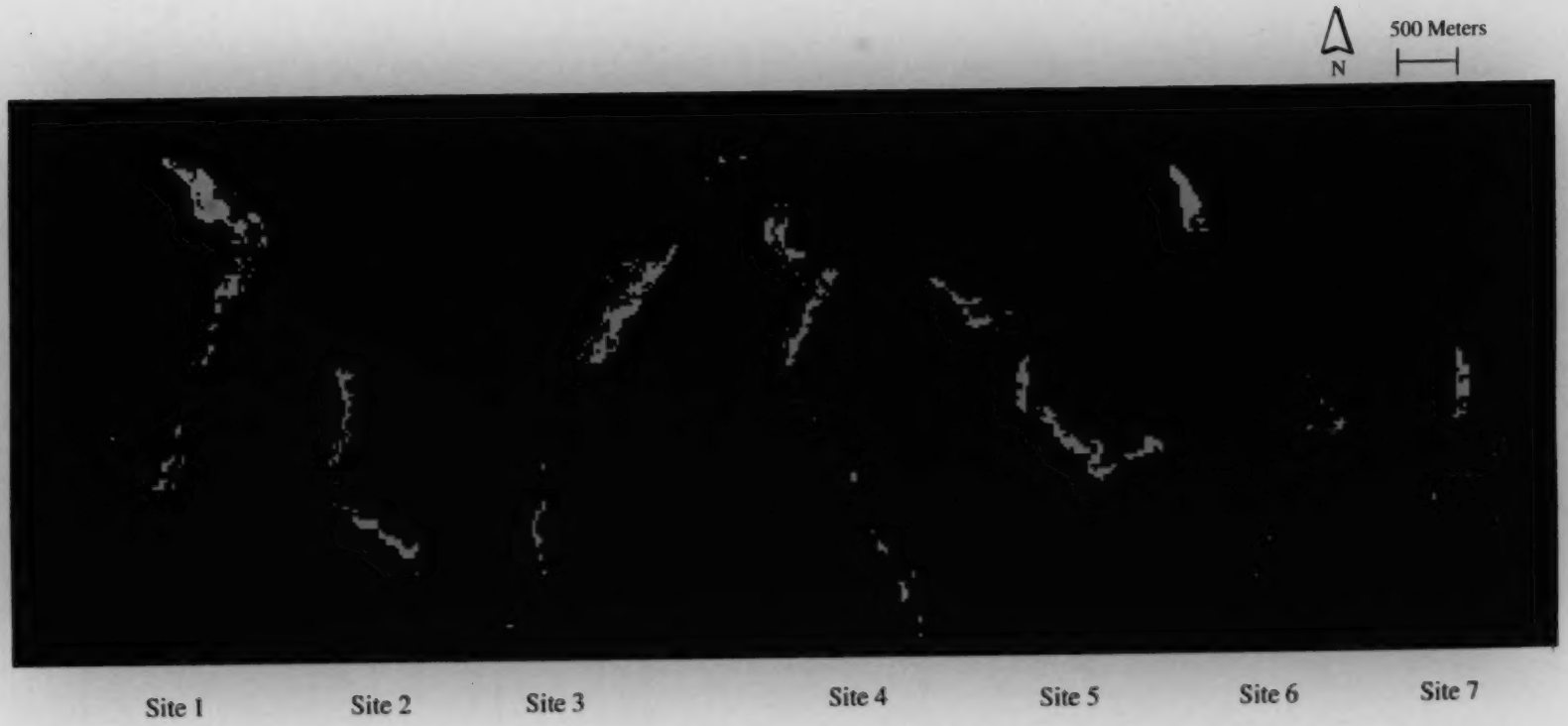

Figure 6.3 1998 ISO classification of selected riparian zone sites, where the grey line indicates the fire boundary, B represents the burned treatment and U the unburned treatment (Sites are in numeric order, not geographically oriented across the landscape).

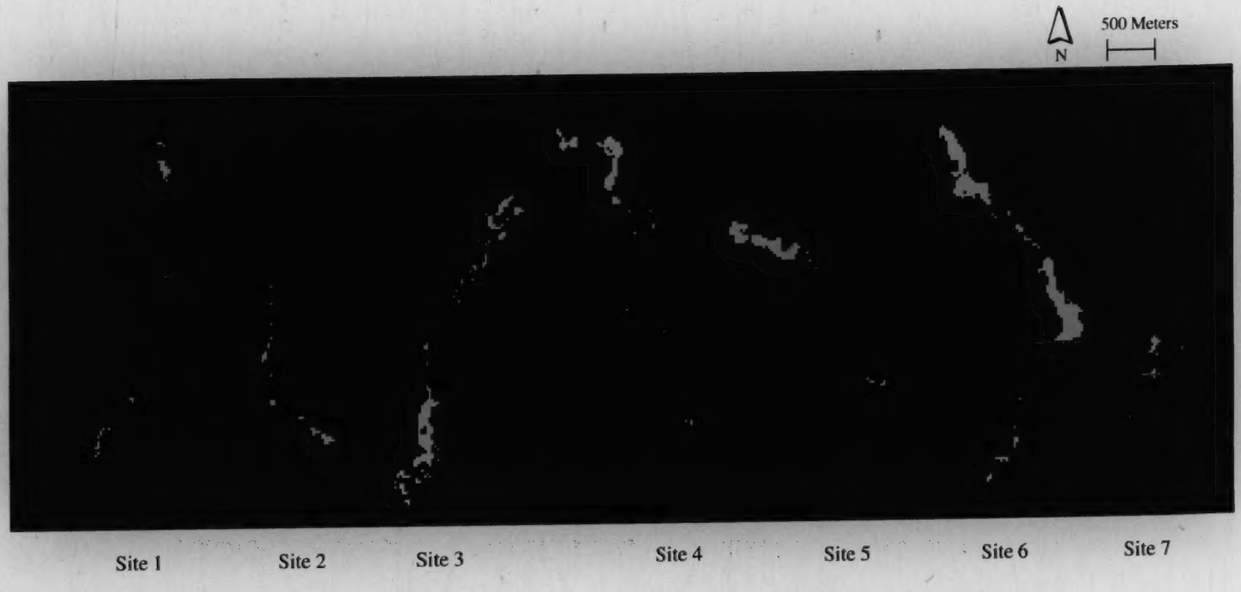

Figure 6.4 2001 ISO classification of selected riparian zone sites, where the grey line indicates the fire boundary, B represents the burned treatment and U the unburned treatment (Sites are in numeric order, not geographically oriented across the landscape).

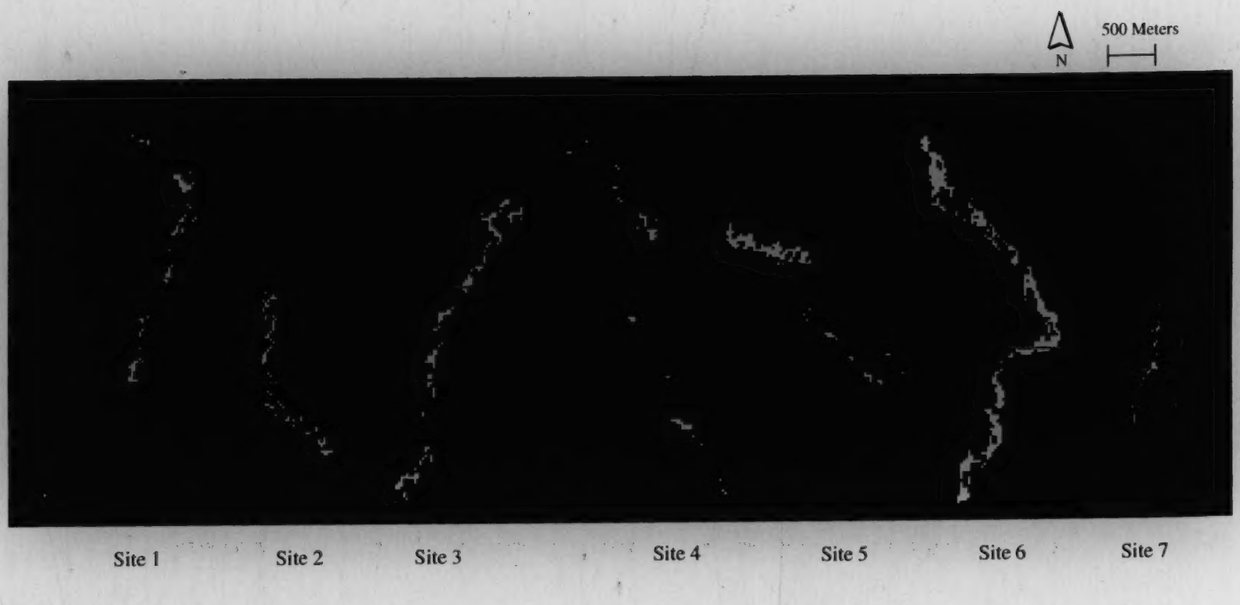

Figure 6.5 2002 ISO classification of selected riparian zone sites, where the grey line indicates the fire boundary, B represents the burned treatment and U the unburned treatment (Sites are in numeric order, not geographically oriented across the landscape).

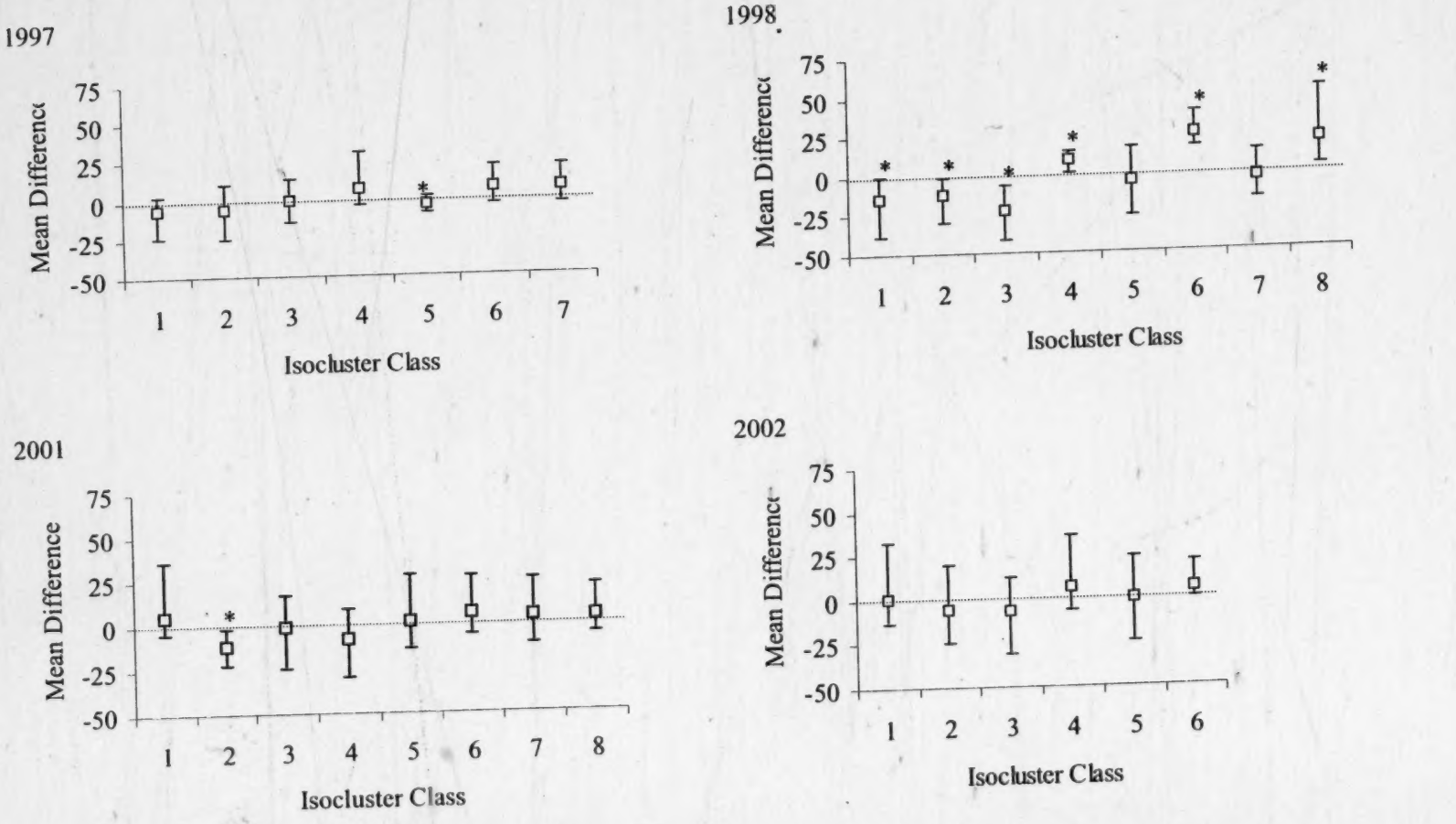

Figure 6.6 Mean difference (unburned – burned) in percent cover of ISO classes, where the dotted line represents zero or no difference, the y-axis error bars represent the minimum and maximum difference observed for each class, and * indicates a significant difference between burned and unburned treatments.

short-shrub, is predominantly found in the low lying areas. Class 7 (yellow) corresponds to the locations of tall shrubs. Classes 2 and 8, composed of field classes from bare soil to lush grass to tall shrubs, are often located along the border of riparian zones where transitions occur to the uplands. In contrast, class 1, similar to classes 2 and 8 in field composition, are located in the valley bottoms of the riparian zones, perhaps representing water, bare soil and shrubs surrounding the channel. In 2002, class 1, representing taller shrubs in addition to lush grass and short shrub-grass, and class 5, moderate to lush grass and short shrub-grass, are distributed in most of the lowlands. The remaining classes are inconsistent in distribution.

Table 6.1 displays the percent of pixels in each ISO spectral class ranked in order of highest to lowest percent cover for each year. The number of classes within each year is slightly different. In the 1997 ISO, 7 classes were identified while in 1998 and 2001, 8 classes were present and in 2002, only 6 (Table 6.1, Figure 5. 5.5). Overall, the distribution of percent of pixels in each class was even in all years.

Table 6.1 Percent of pixels in each ISO class in order of greatest percent cover to lowest.

1997		1998		2001		2002	
Class	Percent	Class	Percent	Class	Percent	Class	Percent
3	16.5	6	17.6	2	16.2	2	21.6
2	15.9	5	13.8	6	15.0	4	19.4
1	15.8	4	12.7	4	14.8	5	18.6
7	15.4	8	12.6	5	14.3	1	16.3
5	14.7	3	12.4	8	13.7	3	14.6
6	11.4	2	11.9	3	10.3	6	9.4
4	10.5	7	9.7	7	8.2		
		1	9.4	1	7.4		

The greatest frequency of significant results observed for the mean difference in percent cover of spectral classes between burned and unburned treatments was in 1998 (Figure 6.6). In 1997 and 2001, a significant difference was observed in only one class, while none were present in 2002. In addition, the deviation of the mean difference in percent cover between treatments was greatest in 1998. However, the variation in the range of difference in percent cover within each class was greatest in 2001 and 2002 then 1997 and least in 1998.

6.4.2 Normalized Difference Vegetation Index Analysis

Figures 6.7 through 6.13 display the NDVI data characterization in frequency histograms for the riparian zones. The average NDVI values at each site (Figure 6.14) indicate the assumed change in land cover between burned and unburned treatments. In 1997, 1998 and 2002 there was a significant difference ($p>0.05$) in average NDVI value between burned and unburned treatments at site 5 (Figures 6.11 and 6.14). Beyond that one site, none of the changes in the average NDVI value were significant or indicated a strong trend between treatments in any year. In 1998, the trend of the average NDVI value in the burned treatment was an increase at sites 1 (Figures 6.7 and 6.14), 2 (Figures 6.8 and 6.14) and 3 (Figures 6.9 and 6.14), a decrease at sites 4 (Figures 6.10 and 6.14) and 5 (Figures 6.11 and 6.14) and no change at sites 6 (Figures 6.12 and 6.14) and 7 (Figures 6.13 and 6.14). In 2001, the main trend in average NDVI value for the burned treatment was a decrease at sites 1 through 4 (Figures 6.7, 6.8, 6.9, 6.10 and 6.14), and an increase at site 5 (Figures 6.11 and 6.14), and no change at sites 6 (Figures 6.12 and 6.14)

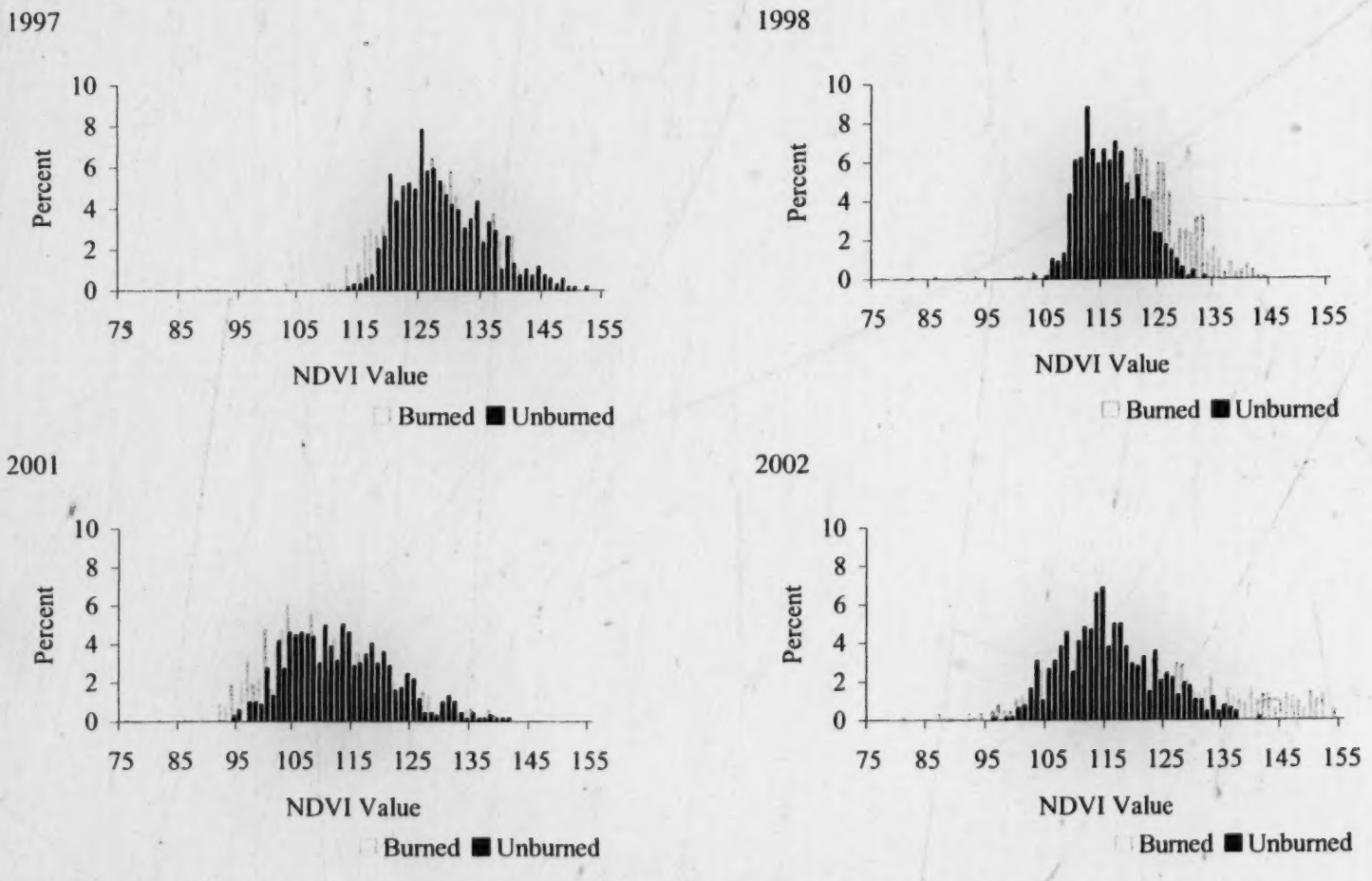

Figure 6.7 Distribution of NDVI values at site 1 for all image dates (based on percent of pixels not absolute number).

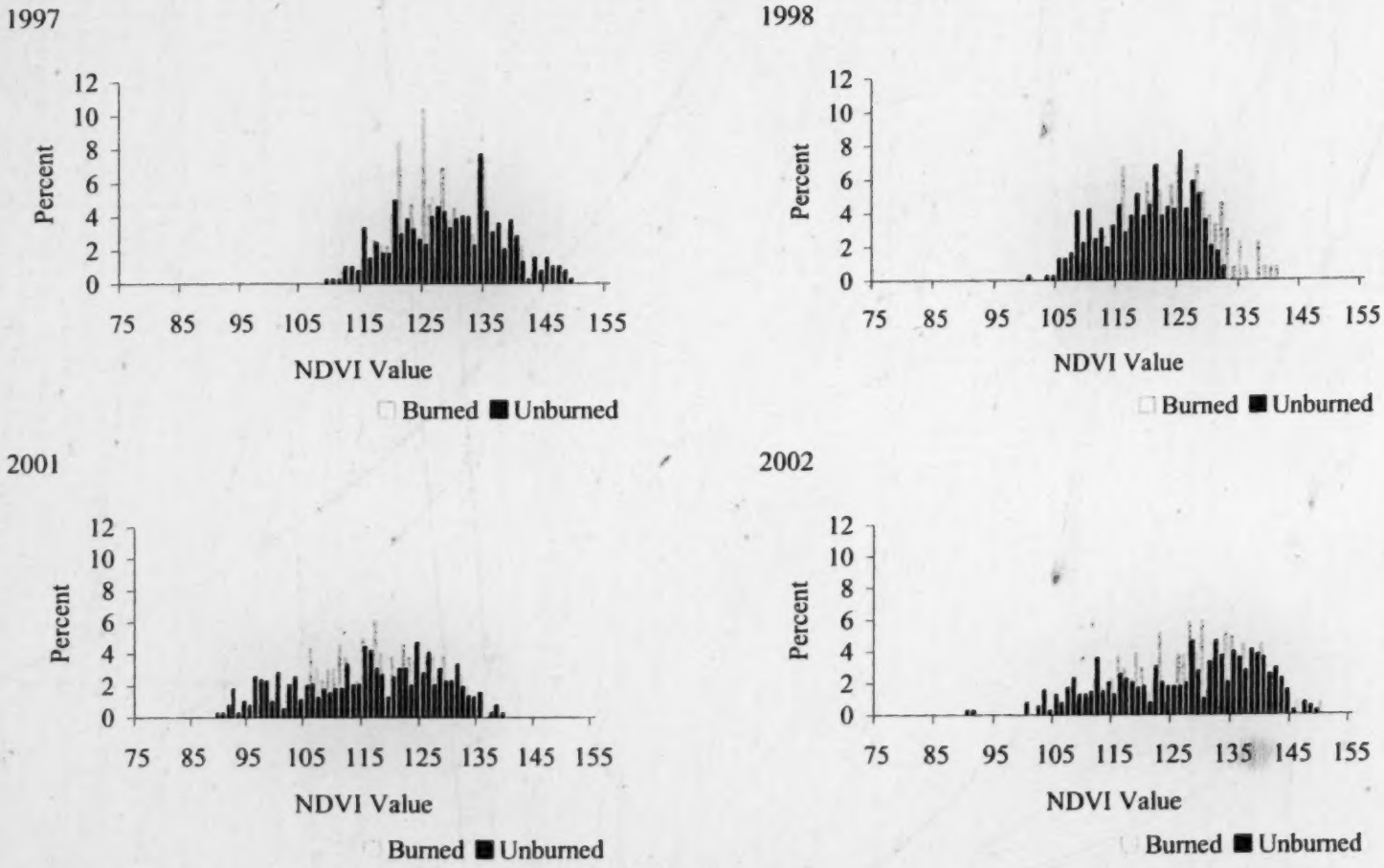

Figure 6.8 Distribution of NDVI values at site 2 for all image dates (based on percent of pixels not absolute number).

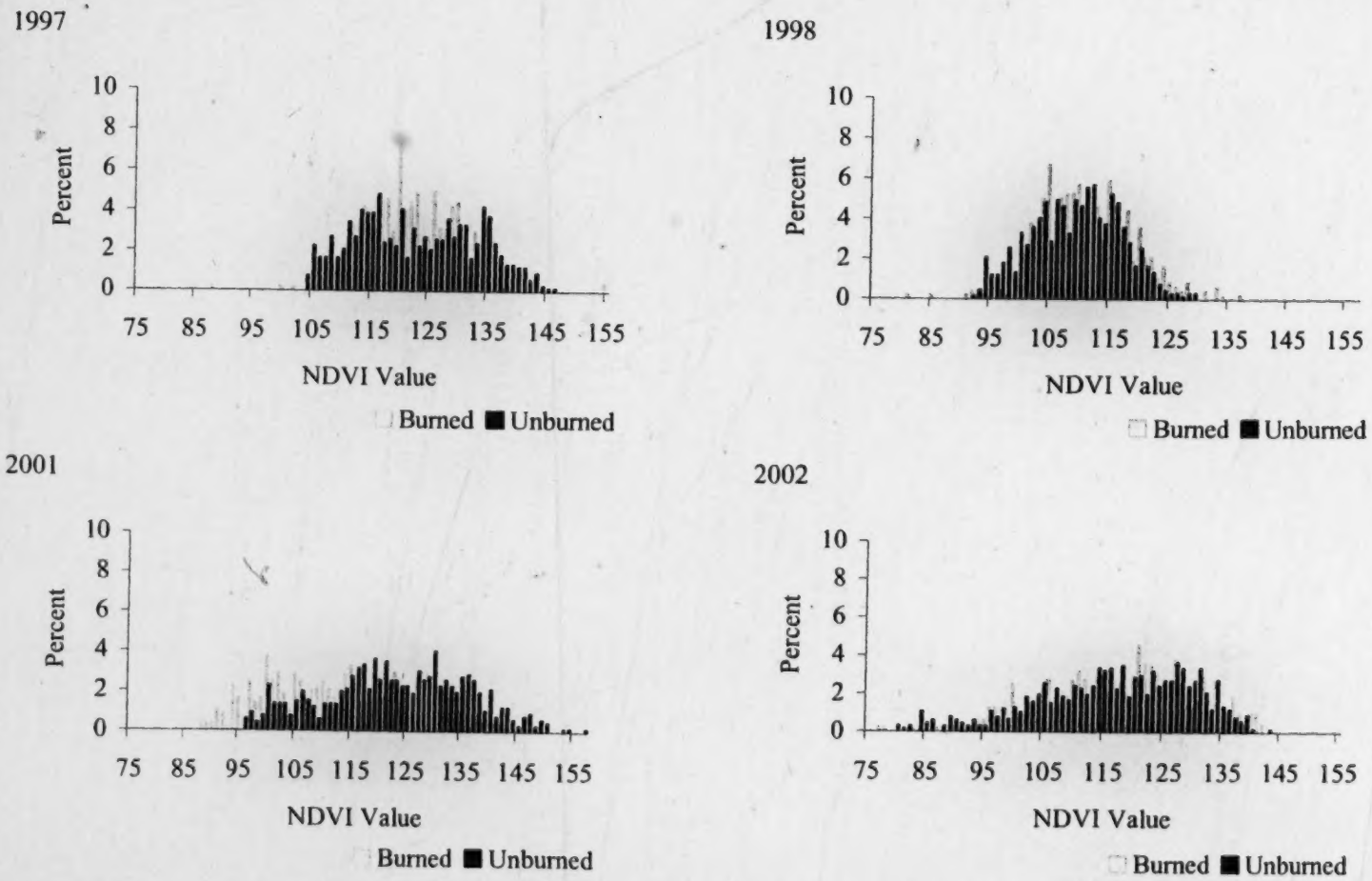

Figure 6. 9 Distribution of NDVI values at site 3 for all image dates (based on percent of pixels not absolute number).

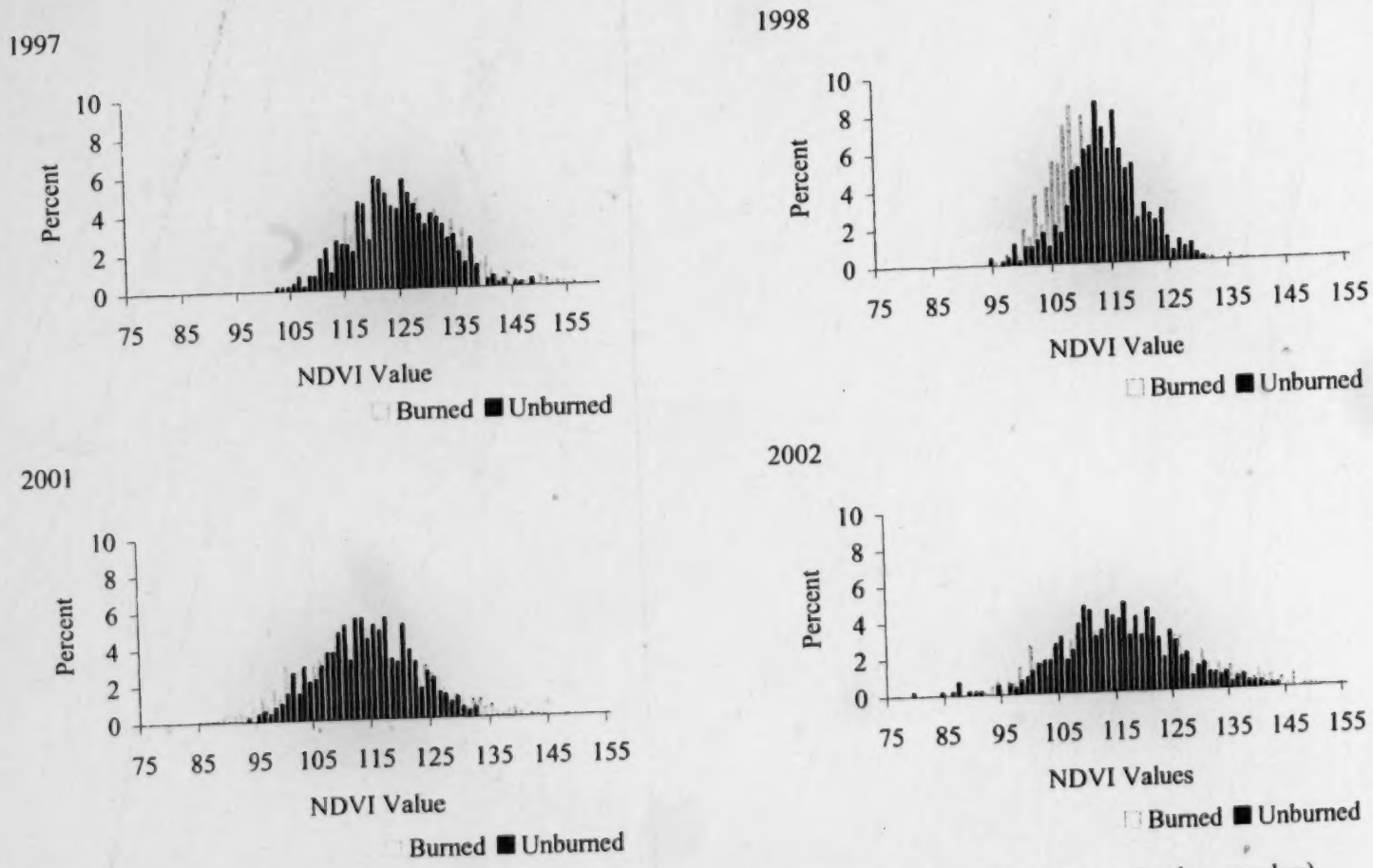

Figure 6.10 Distribution of NDVI values at site 4 for all image dates (based on percent of pixels not absolute number).

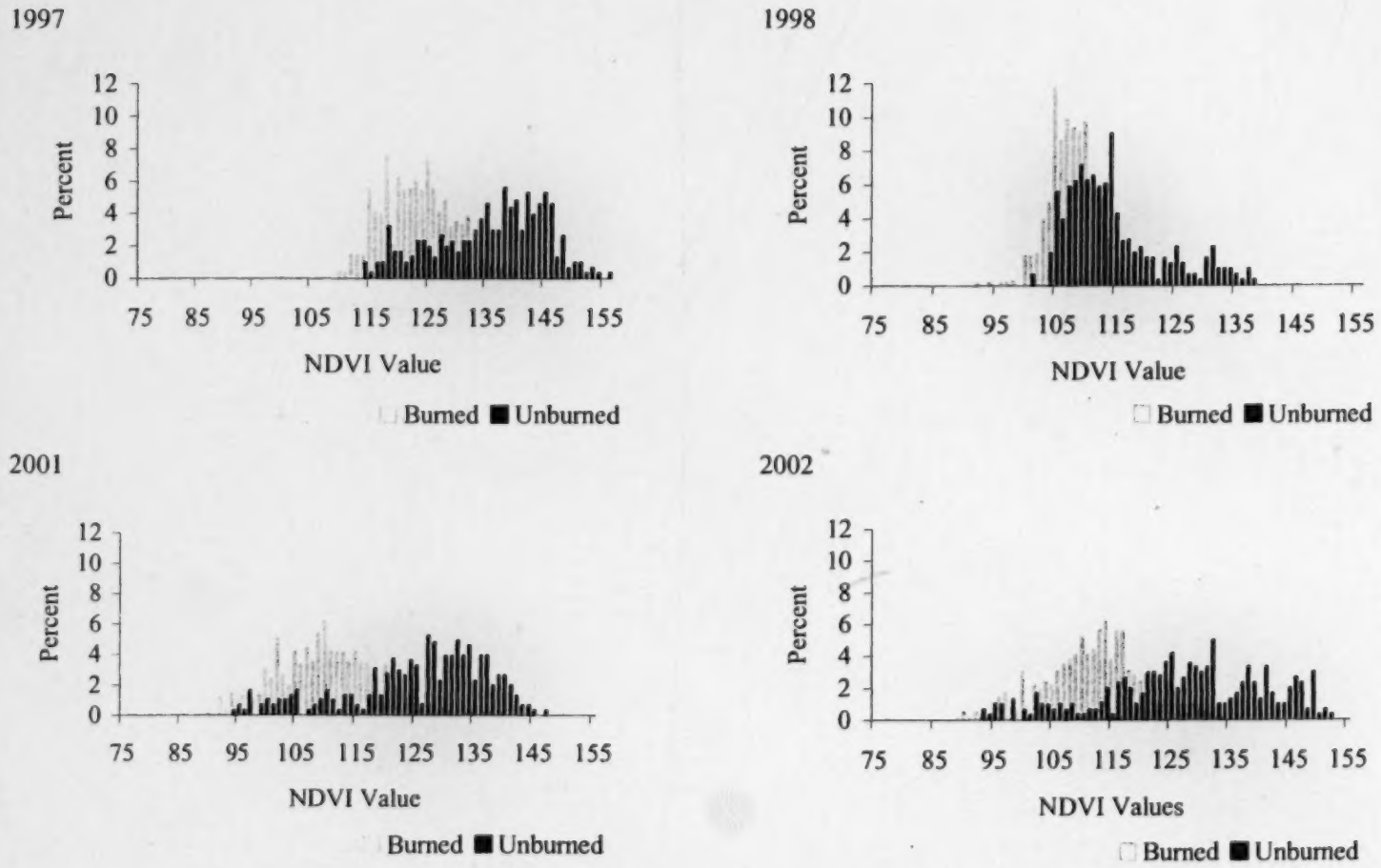

Figure 6.11 Distribution of NDVI values at site 5 for all image dates (based on percent of pixels not absolute number).

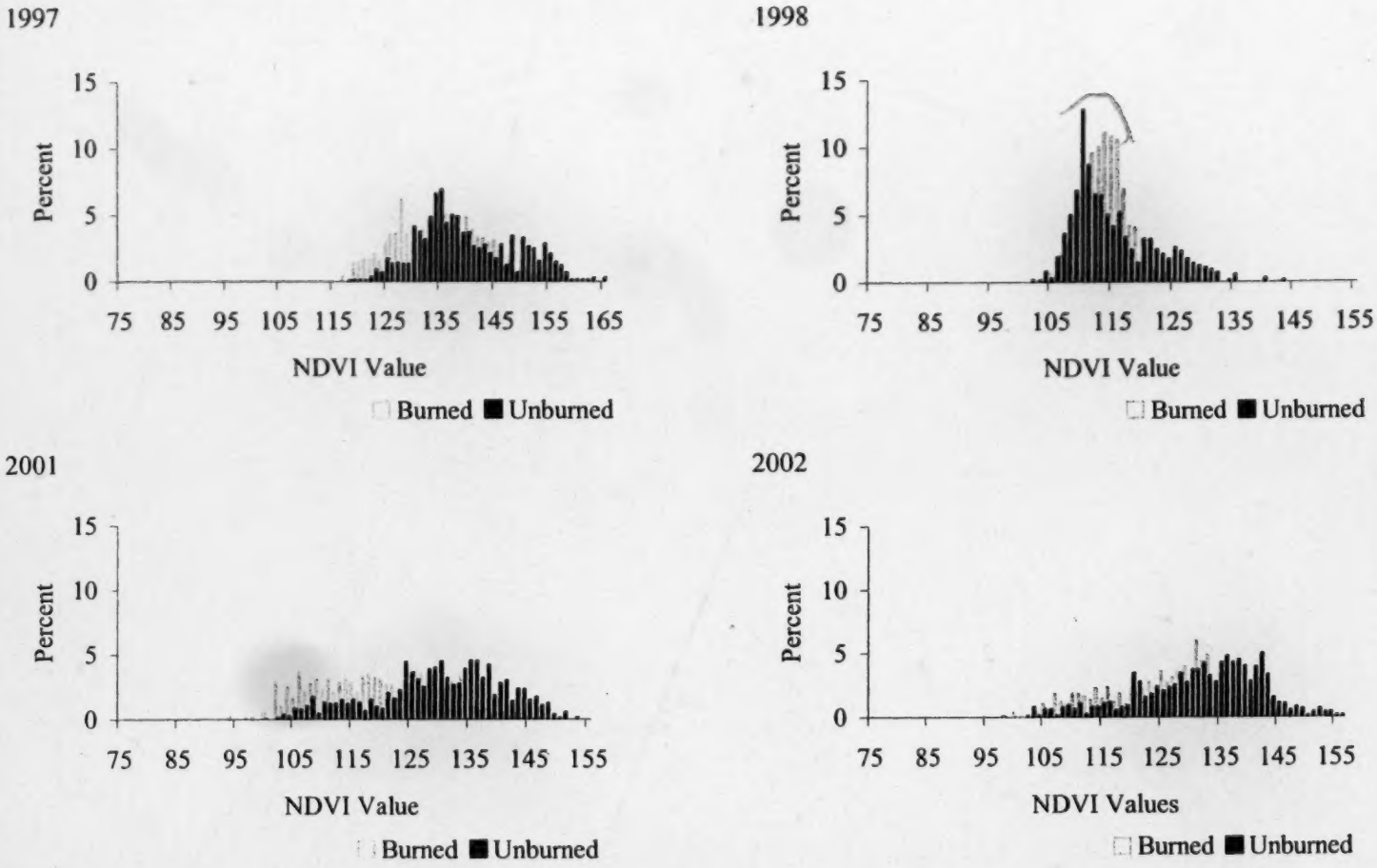

Figure 6.12 Distribution of NDVI values at site 6 for all image dates (based on percent of pixels not absolute number).

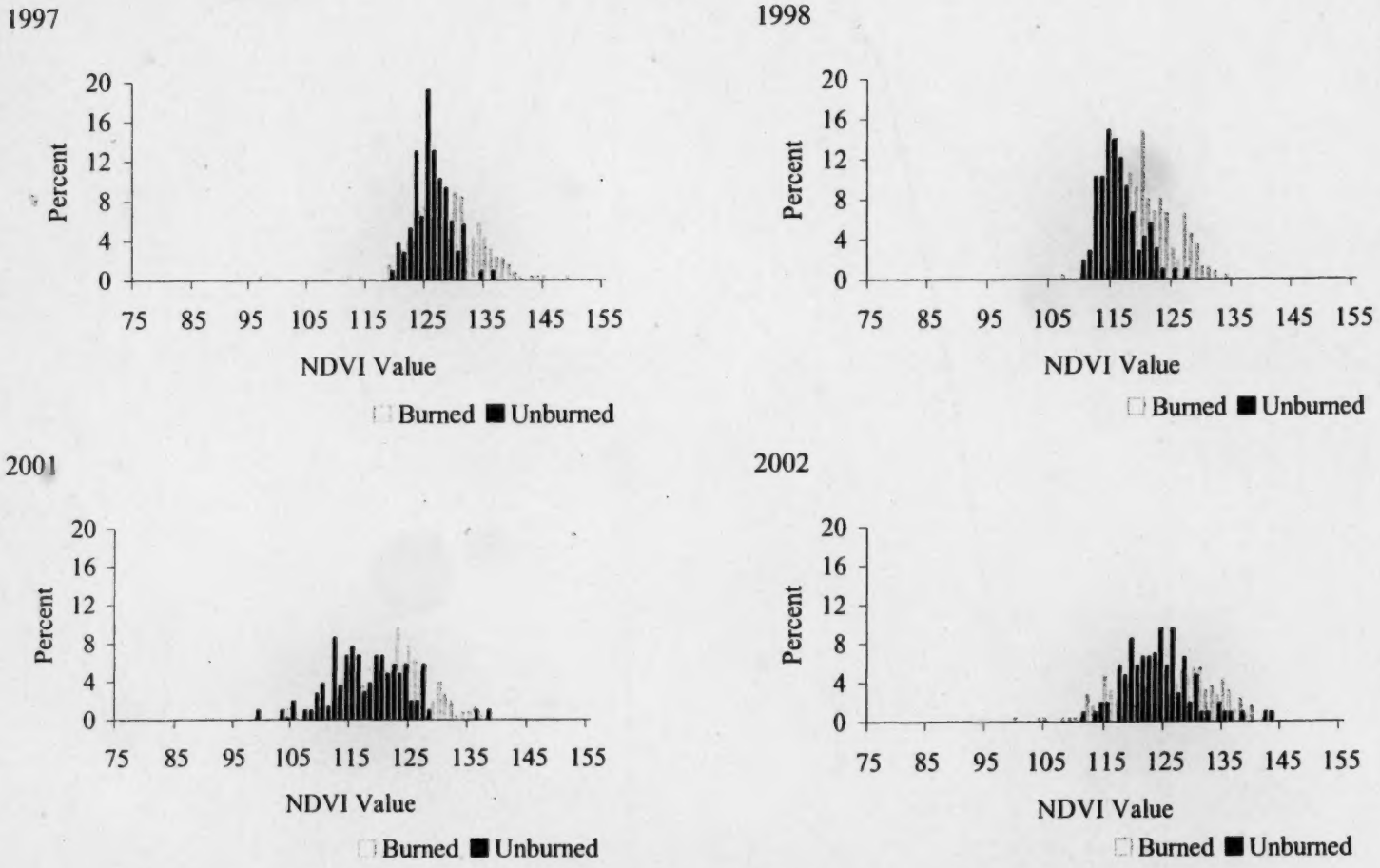

Figure 6.13 Distribution of NDVI values at site 7 for all image dates (based on percent of pixels not absolute number).

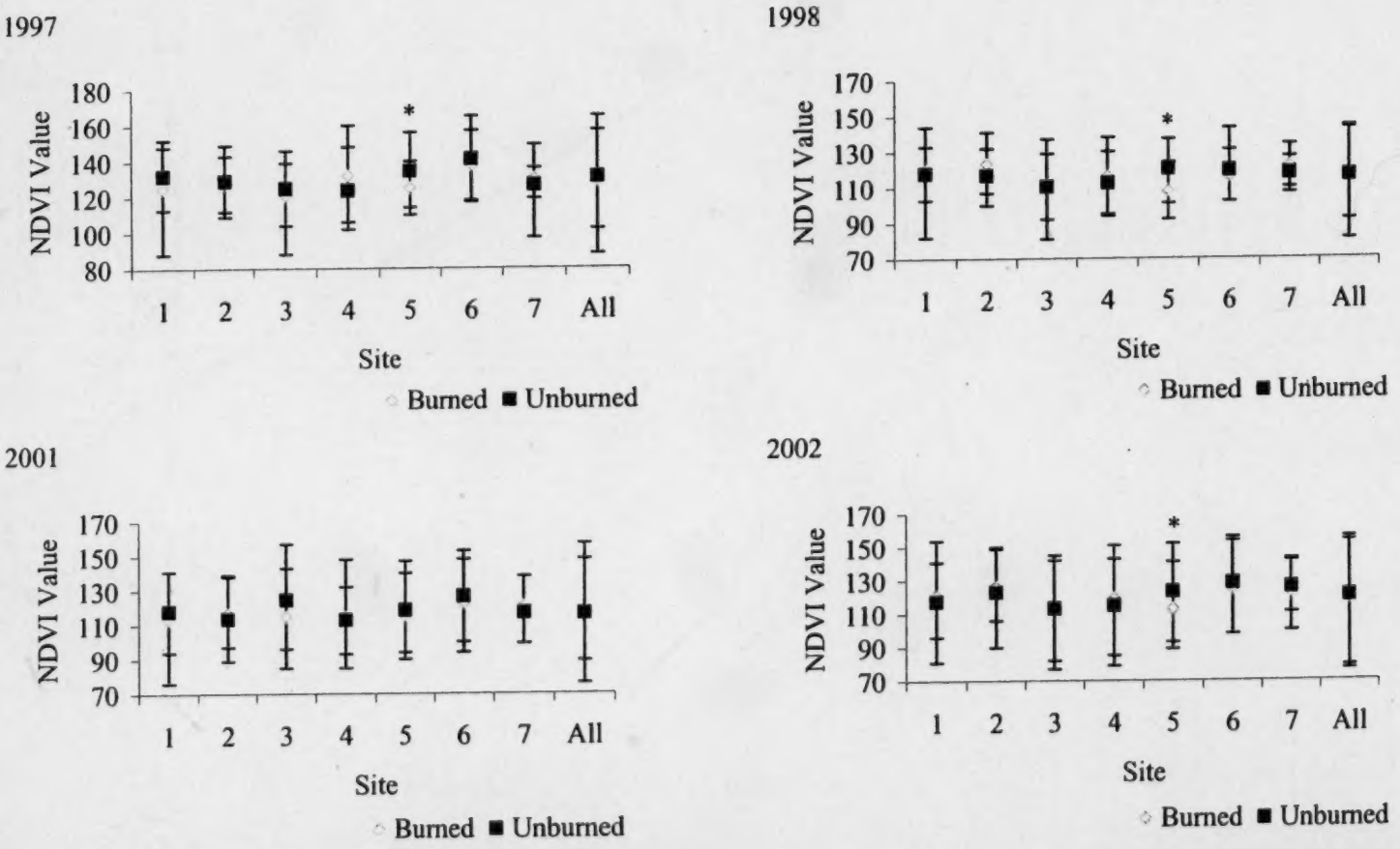

Figure 6.14 Average NDVI value for burned and unburned treatments across the riparian zones, where the * indicates a significant difference between the burned and unburned NDVI values and the y-axis error bars represent the minimum and maximum NDVI value in each site.

and 7 (Figures 6.13 and 6.14). In 2002, the average NDVI value for the burned treatment increased at sites 1 (Figures 6.7 and 6.14), 3 (Figures 6.9 and 6.14) and 4 (Figures 6.10 and 6.14), decreased in sites 5 (Figures 6.11 and 6.14) and 7 (Figures 6.13 and 6.14), and did not change for sites 2 (Figures 6.8 and 6.14) and 6 (Figures 6.12 and 6.14).

Taken together, there were no significant differences between burned and unburned treatments. In 1997 the average NDVI value was less for the burned treatment than the unburned. In 1998, the average NDVI value for the burned treatment increased slightly resulting in equal burned and unburned treatment values. The average NDVI values for burned and unburned treatments remained equal in 2001 and 2002.

6.4.3 Principal Component Analysis

Figures 6.15 through 6.18 display the PCA data characterization for the riparian zones. Given that class numbers within each year are arbitrarily assigned, colours and class numbers cannot be directly compared between years. Table 6.2 displays the percent of pixels for all sites combined in each PCA spectral class ranked in order of highest to lowest percent cover for each year. The distribution of the pixels was fairly equal between classes in 1997, 2001 and 2002.

The distribution of PCA spectral classes in 2001 and 2002 are illustrated in Figures 6.17 and 6.18 with colours defined in Table 6.3. In 2001, classes 1 (white), 2 (red), 3 (green), 6 (magenta) and 7 (yellow) are all lush grass, short shrub-grass and shrub vegetation located mainly in the valley bottoms of the riparian zones, but also occur near the boundary. Classes 4 (blue), 8 (orange), and 9 (chartreuse) are short and moderate shrub-grass communities in the uplands. In 2002, the class distribution trends are less distinct. Classes 3 (green), moderate to lush grass, and short shrub-grass, and 5 (cyan),

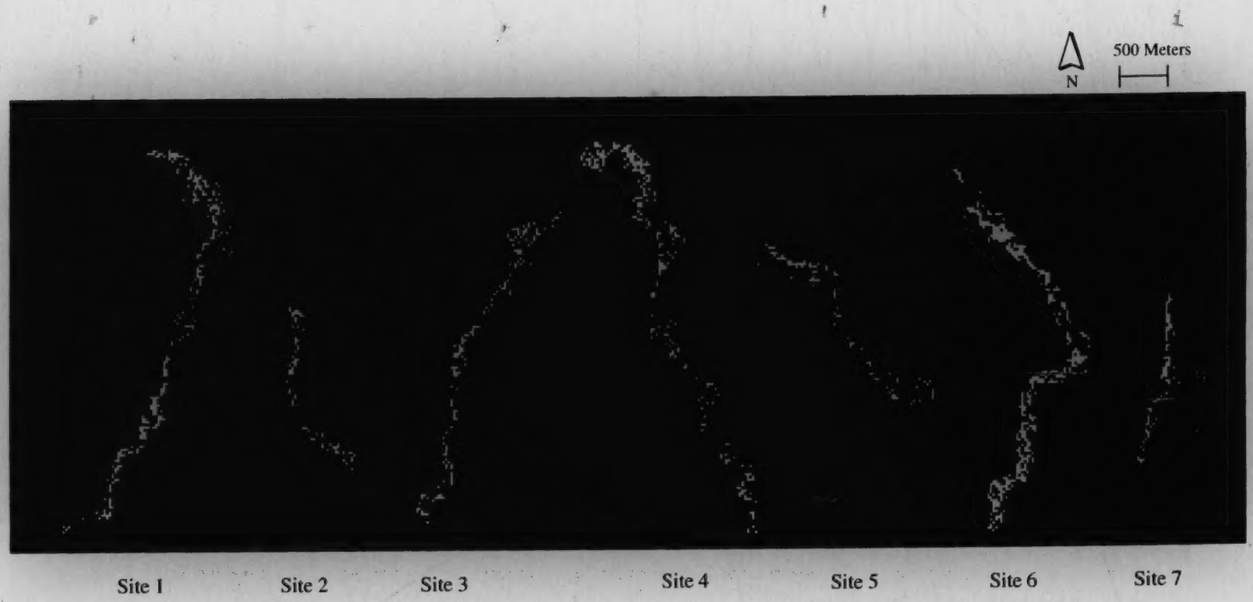

Figure 6.15 1997 PCA classification of selected riparian zone sites, where the grey line indicates the fire boundary, B represents the burned treatment and U the unburned treatment (Sites are in numeric order, not geographically oriented across the landscape).

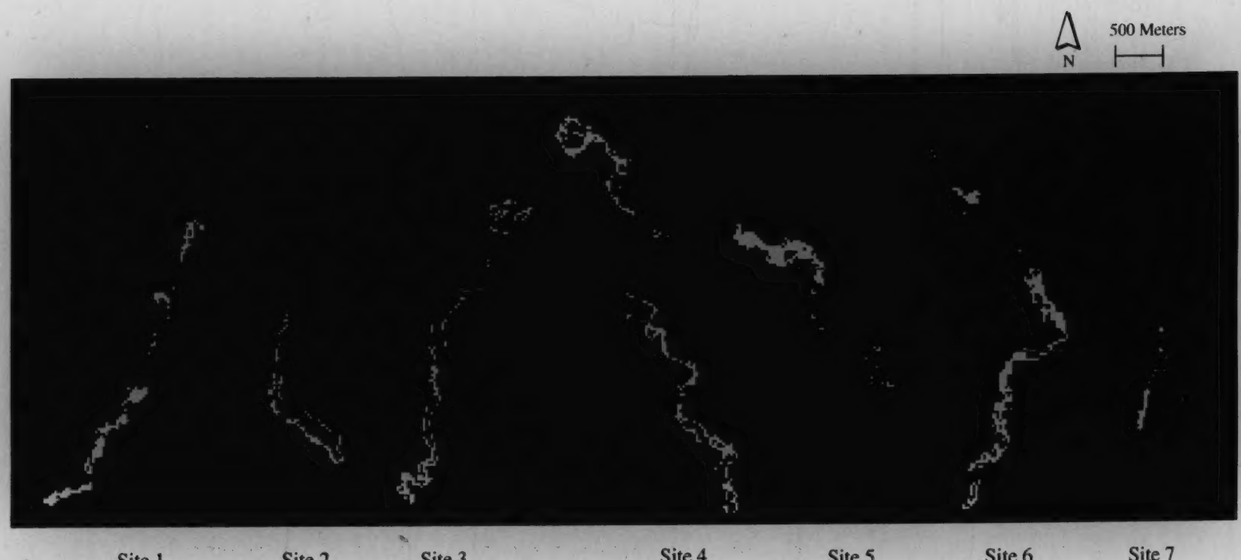

Figure 6.16 1998 PCA classification of selected riparian zone sites, where the grey line indicates the fire boundary, B represents the burned treatment and U the unburned treatment (Sites are in numeric order, not geographically oriented across the landscape).

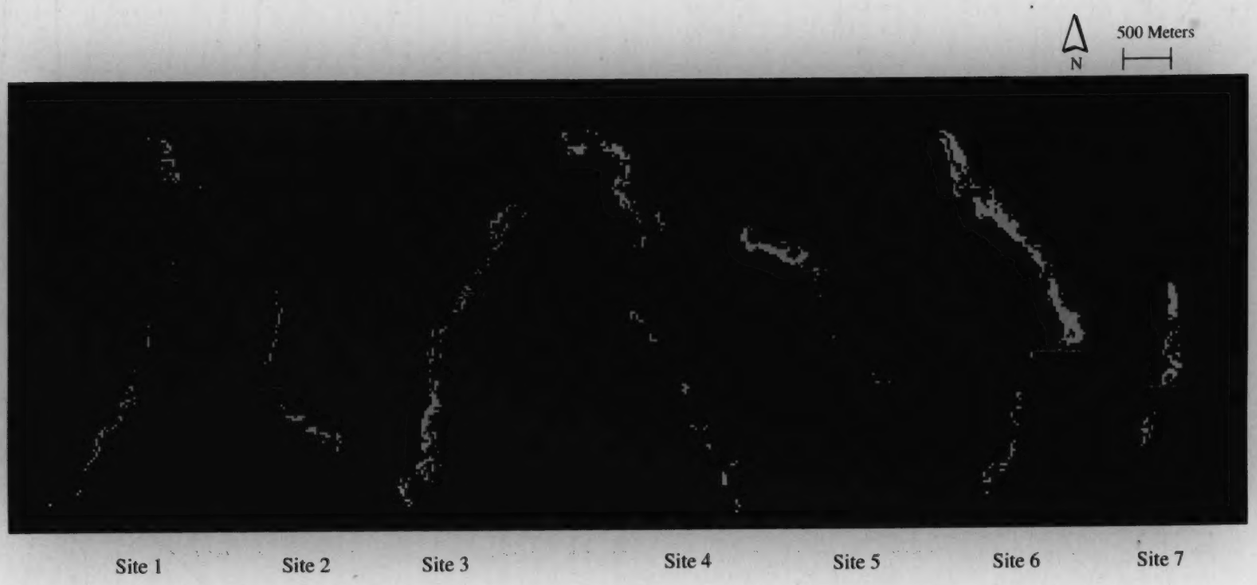

Figure 6.17 2001 PCA classification of selected riparian zone sites, where the grey line indicates the fire boundary, B represents the burned treatment and U the unburned treatment (Sites are in numeric order, not geographically oriented across the landscape).

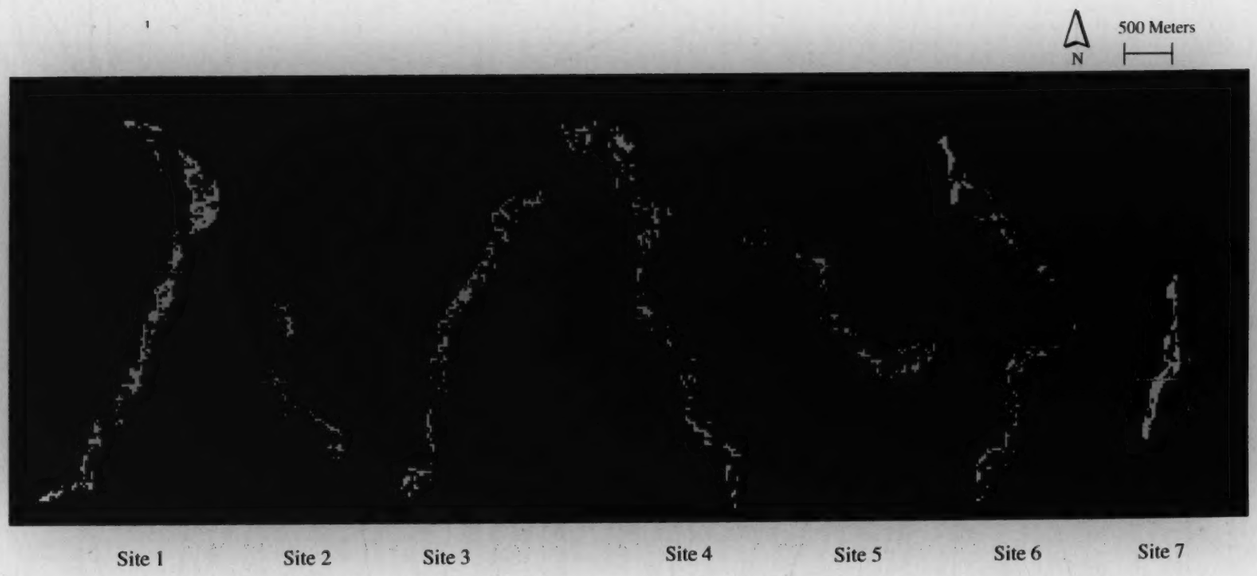

Figure 6.18 2002 PCA classification of selected riparian zone sites, where the grey line indicates the fire boundary, B represents the burned treatment and U the unburned treatment (Sites are in numeric order, not geographically oriented across the landscape).

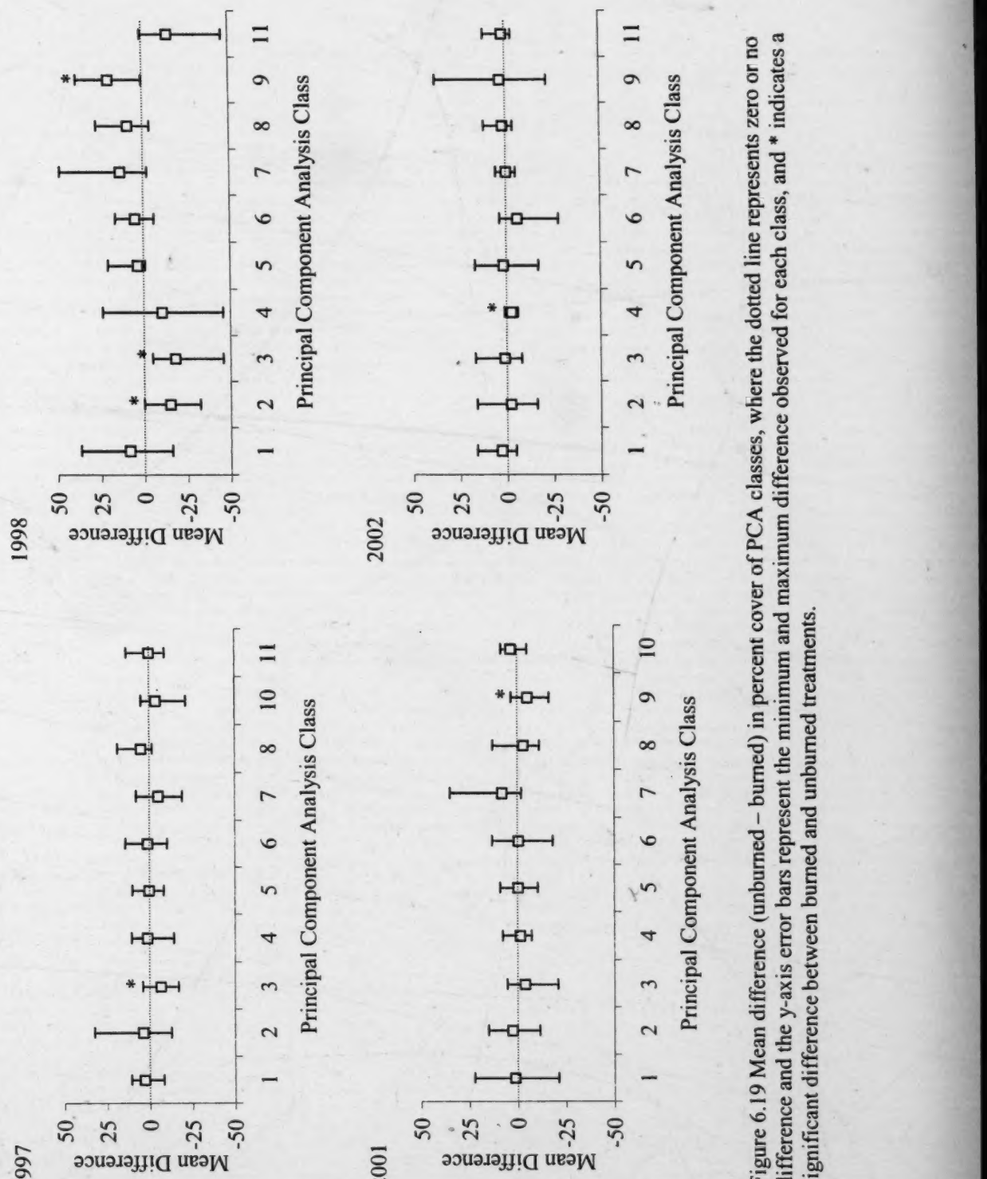

Figure 6.19 Mean difference (unburned – burned) in percent cover of PCA classes, where the dotted line represents zero or no difference and the y-axis error bars represent the minimum and maximum difference observed for each class, and * indicates a significant difference between burned and unburned treatments.

moderate to lush grass, short shrub-grass and shrubs, occur mainly in the lowlands. Class 8 (orange), grass and bare soil occur in the uplands near the edge of the riparian zone. The remaining classes are inconsistently distributed in the lowland and upland areas of the riparian zone.

Table 6.2 Percent of pixels in each PCA spectral class in order of greatest percent cover to lowest.

1997		1998		2001		2002	
Class	Percent	Class	Percent	Class	Percent	Class	Percent
10	11.8	9	13.7	10	11.6	2	11.8
1	11.6	1	12.1	5	11.4	7	11.7
5	11.3	2	11.1	1	11.3	6	11.5
7	10.7	8	11.1	9	10.7	1	11.2
3	10.6	4	11.0	4	10.7	4	10.5
11	10.5	3	10.9	6	10.3	3	10.0
6	10.3	7	10.7	2	9.7	11	9.4
4	10.0	6	8.5	3	9.6	9	9.1
2	8.1	11	7.9	8	8.2	5	8.6
8	5.2	5	3.0	7	6.5	8	6.3

The paired t-test results are displayed in Figure 6.19 for all four image dates. In 1997, 2001 and 2002, there is one class with a significant result for the mean difference in percent cover between burned and unburned treatments, while in 1998, there are 3 classes with significant results. The deviation of the mean difference of percent cover from zero and the variation in mean difference of percent cover within each class is considerably greater in 1998 than any other year.

6.4.4 Patch Analysis for Isocluster and Principal Component Analysis Characterizations

Patch statistics between burned and unburned areas varied considerably (Figure 6.20 and Tables 6.4 and 6.5). The trends in number of patches in burned and unburned treatments were similar. For both the ISO and PCA classifications, the number of patches was high in 1997, dropped considerably in 1998, then increased in 2001 (Figure 6.20 and Table 6.4 and 6.5). In 2002, the number of patches remained similar to 2001 for the PCA classification and dropped for the ISO classification. Figure 6.21 displays the decrease in patch size as the number of patches increases. In 1997, 2001 and 2002 the average patch sizes were considerably smaller than in 1998 for the PCA data. For the ISO data, the average patch size was very small in 1997 and 2001, but increased in 1998 and 2002. In the ISO (Table 6.4) and PCA (Table 6.5) data, dominant classes were not restricted to either burned or unburned area, but were present in both. In 1998, fewer dominant classes occurred in both the burned and unburned areas of each site, compared to 1997, 2001 and 2002.

A visual assessment of the distribution of dominant classes in the ISO classification is displayed in Figures 6.2 to 6.5, highlights of site differences between treatments are displayed in Table 6.4, and the colour renditions linking the table and the figures are described in Table 6.3. In 1998, the fire line separating burned and unburned areas is distinct due to different dominant spectral classes. The pattern of spectral class distribution in 2001 compared to 1997 for sites 1, 3, 6 and 7 is similar, for site 2 is not consistent, and for 4 and 5 is less patchy. The dominant cover in burned and unburned areas in 2002 compared to 1997 is similar at sites 1, 2, 4, 6, and 7. Site 3 has different dominant classes but similar minor classes. Site 5 is very different in 1997 and 2002.

A visual assessment of the distribution of dominant classes in the PCA classification is displayed in Figures 6.15 to 6.18, highlights of site differences between treatments are displayed in Table 6.5, and the colour renditions linking the table and the figures are described in Table 6.3. The PCA patches are smaller and more isolated than those in the ISO classification. In 1998, the fire line separating burned and unburned areas is distinct in all sites except site 7 due to different dominant spectral classes. The pattern of spectral class distribution in 2002 compared to 1997 for sites 1 to 7 is similar, but less patchy. Dominant cover in burned and unburned areas in 2002 compared to 1997 is similar at sites 1, 3, 4, and 7, mainly due to lack of spectral class dominance. Sites 2 and 6 have different dominant classes but similar minor classes. Site 5 has very different burned and unburned spectral classes in each year. Therefore, the PCA and ISO classifications did not detect distinct trends of changes in spectral class patches.

Table 6.3 Colour renditions associated with ISO and PCA spectral classes.

Spectral Class Number	Spectral Class Colour
1	White
2	Red
3	Green
4	Blue
5	Cyan
6	Magenta
7	Yellow
8	Orange
9	Pale Green
10	Chartreuse
11	Navy

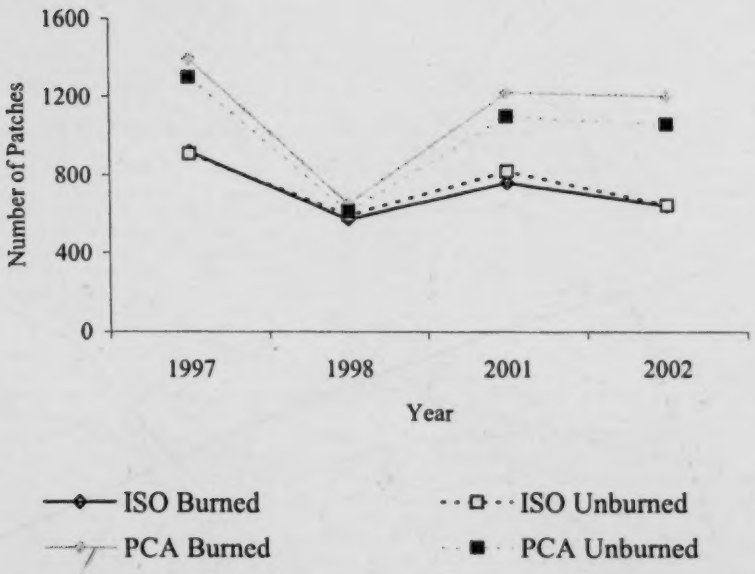

Figure 6.20 Number of patches in burned and unburned treatments observed across the riparian zones in each image date.

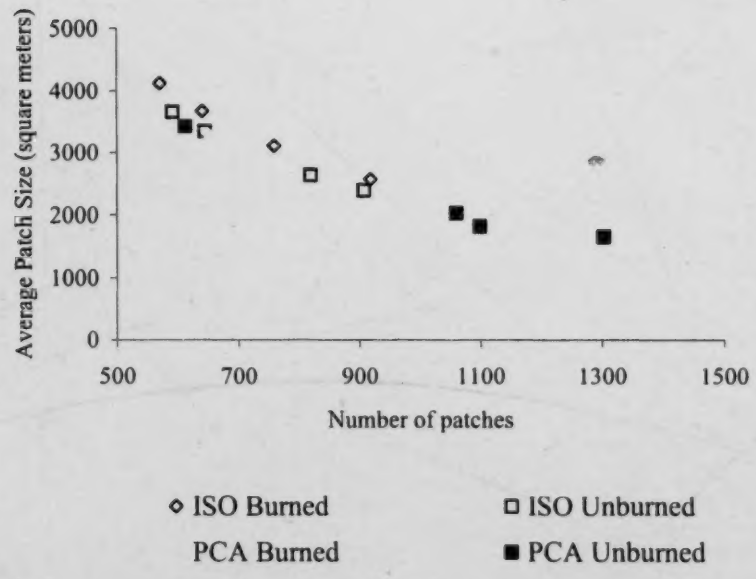

Figure 6.21 Decrease in average patch size within burned and unburned treatments as the number of patches observed across the riparian zones increase.

Table 6.4 Total number of patches, average patch size and dominant class cover (20% or greater) observed with the ISO classification for burned and unburned treatments in each riparian zone for all image dates. Dominant class numbers occurring in both burned and unburned areas of one site are bold

Site			Year			
			1997	1998	2001	2002
1	Burned	Total # patches	106	79	106	116
		Average area (m^2)	3970	5327	3970	3613
		Dom. Classes	5, **7**	3, 7	**2**	**2**, 5
	Unburned	Total # patches	217	108	217	101
		Average area (m^2)	2009	4020	2009	4241
		Dom. Classes	**7**	6	**2**, 4, 6	**2**, 3
2	Burned	Total # patches	61	50	61	46
		Average area (m^2)	1356	1654	1356	1841
		Dom. Classes	2, 5	5, **7**	4, **6**	1, 2, **5**
	Unburned	Total # patches	123	91	123	101
		Average area (m^2)	1995	2710	1995	2435
		Dom. Classes	no dominant	6, **7**	5, **6**, 8	**5**
3	Burned	Total # patches	132	74	132	92
		Average area (m^2)	2254	4005	2254	3205
		Dom. Classes	1, 2	1, 3	2, 5	2, 3
	Unburned	Total # patches	159	72	159	152
		Average area (m^2)	2590	5788	2590	2715
		Dom. Classes	6	8	7	4, 6
4	Burned	Total # patches	176	160	176	189
		Average area (m^2)	3981	4409	3981	3704
		Dom. Classes	1, **2**	2	**8**	**4**, **6**
	Unburned	Total # patches	135	116	135	135
		Average area (m^2)	2551	2975	2551	2606
		Dom. Classes	1, **2**, 3	4, 6, 8	6, **8**	2, **4**, **6**
5	Burned	Total # patches	115	76	115	99
		Average area (m^2)	3451	5179	3451	3972
		Dom. Classes	1, 2	1, **2**	2, 4	2, 3
	Unburned	Total # patches	58	87	58	40
		Average area (m^2)	3298	2193	3298	4701
		Dom. Classes	4	**2**	3	1, 5
6	Burned	Total # patches	120	87	120	68
		Average area (m^2)	2498	3392	2498	4442
		Dom. Classes	3, **4**	3, 5	3, 4	**1**, 2
	Unburned	Total # patches	110	97	110	101
		Average area (m^2)	4247	4831	4247	4651
		Dom. Classes	**4**, 7	6, 8	1, 5, 7	**1**, 5
7	Burned	Total # Patches	49	45	49	31
		Average Area (m^2)	3328	3651	3328	5178
		Dom. Classes	**5**, 7	3, **5**	3, **5**	**4**, 5
	Unburned	Total # Patches	17	21	17	15
		Average Area (m^2)	3864	3218	3864	4382
		Dom. Classes	**5**, 6	**5**, 6	**5**, 6	**4**

Table 6.5 Total number of patches, average patch size and dominant class cover (20% or greater) observed with the PCA classification for burned and unburned treatments in each riparian zone for all image dates. Dominant class numbers occurring in both burned and unburned areas of one site are bold

			Year			
Site			1997	1998	2001	2002
1	Burned	Total # patches	258	93	179	177
		Average area (m^2)	1600	4478	2227	2357
		Dom. Classes	**11**	3, 11	no dominant	no dominant
	Unburned	Total # patches	267	82	279	174
		Average area (m^2)	1616	5232	1428	2462
		Dom. Classes	**11**	1, 9	4	1, 3
2	Burned	Total # patches	72	54	76	67
		Average area (m^2)	1169	1532	989	1264
		Dom. Classes	no dominant	**6**, 11	5	2
	Unburned	Total # patches	197	97	174	148
		Average area (m^2)	1245	2522	1318	1662
		Dom. Classes	2	**6**, 8	no dominant	5
3	Burned	Total # patches	214	130	220	201
		Average area (m^2)	1364	2275	1266	1458
		Dom. Classes	3, **5**	2	**no dominant**	**no dominant**
	Unburned	Total # patches	277	130	192	258
		Average area (m^2)	1509	3199	2008	1599
		Dom. Classes	**5**	5, 7, 9	**no dominant**	**no dominant**
4	Burned	Total # patches	299	166	299	299
		Average area (m^2)	1712	3351	1863	1877
		Dom. Classes	3	2, 4	**no dominant**	**no dominant**
	Unburned	Total # patches	279	166	208	212
		Average area (m^2)	1237	2077	1534	1660
		Dom. Classes	no dominant	1, 9	**no dominant**	**no dominant**
5	Burned	Total # patches	271	75	199	209
		Average area (m^2)	1454	5247	1850	1881
		Dom. Classes	3, 4	2, 4	3	6
	Unburned	Total # patches	91	47	73	55
		Average area (m^2)	2086	4045	2445	3419
		Dom. Classes	2	7	6, 7	9
6	Burned	Total # patches	202	109	183	193
		Average area (m^2)	1476	2258	1523	1565
		Dom. Classes	**1**	1	6	9
	Unburned	Total # patches	146	71	138	195
		Average area (m^2)	3211	5800	3160	2409
		Dom. Classes	**1**, 8, 10	9	1, 2, 7	2, 5
7	Burned	Total # Patches	79	26	63	56
		Average Area (m^2)	2068	5748	2337	2867
		Dom. Classes	no dominant	3, 11	1	**1**, 2, **7**
	Unburned	Total # Patches	46	20	35	18
		Average Area (m^2)	1462	3360	1691	3651
		Dom. Classes	7	1, 4, 9	5, 10	**1**, **7**

6.5 Discussion

6.5.1 Isocluster Classification

The number of classes derived using the ISO is variable depending upon the data diversity and can therefore be used to infer trends in the data. The number of MLC spectral classes increased immediately following the fire and then decreased in 2002 suggesting that the distribution of vegetation in 1998 and 2001 was more complex than in either 1997 or 2002. These observations support the expected trend of greater difference of land cover between burned and unburned treatments immediately following the fire, then a decrease in deviation over time.

The increase in number of significant differences and the higher deviation of the mean difference of percent cover between burned and unburned treatments in the 1998 image data also support the higher expected difference in land cover between the two treatments. In addition to the decrease in spectral complexity in 2002, the expected decrease in difference between burned and unburned treatment over time is observed by the decrease in deviation of mean difference in percent cover of ISO spectral classes from zero in 2001 and 2002. The significant difference observed in 1997 is background variation suggesting that by 2001, the burned and unburned areas of the selected riparian zones have returned to pre-fire condition.

6.5.2 Normalized Difference Vegetation Index

The lack of a strong trend and significant differences in average NDVI value between burned and unburned treatments over the 4 image dates does not reflect the

expected changes in the burned community over time. Salvador *et al.* (2000) observed in the Mediterranean that if vegetation regrowth following fire is successful, then the effect of the fire may not be detectable using NDVI, a finding supported by the results here. Furthermore, Oleson *et al.* (1995) found considerable overlap between burned and unburned NDVI values following fire. This is not surprising given that the NDVI is an index of greenness and is not necessarily species specific.

The increased in NDVI value in 1998 and 2002 indicated greater percent cover of vegetation, perhaps due to regrowth after the fire, a decrease in litter and greater precipitation. The significant differences observed at site 5 for both pre- and post-fire dates, indicate that other factors besides the fire are influencing vegetation cover. The decrease in NDVI value in 2001 indicates a greater percent cover of soil, which may be a result of fire, drought, or grazing pressure.

6.5.3 Principal Component Analysis

The number of classes identified in the PCA was fixed, thus changes to the number of classes identified each year was not a factor used to compare complexity of the spectral class data within each image dataset. However, the distribution of pixels within classes was assessed. The distributions of pixels in 1997, 2001 and 2002 were very similar to each other. In contrast, the distribution of pixels in 1998 showed a higher proportion of pixels in the dominant class, while the least dominant class had considerably fewer than the other three years of data. These differences found in 1998 support the expectation of greater relative deviation of the burned treatment from the

unburned treatment immediately after the fire, while the similarity of 2001and 2002 to the 1997 data point towards a decrease in deviation with increased time after fire.

The paired t-test also supported the expected trend in landscape changes following fire. Immediately following the fire the deviation increased, as indicated by more significant differences in the 1998 image data. In contrast, the 1997, 2001 and 2002 results suggest that by 2001 and 2002 the landscape differences had returned to levels similar to those in 1997.

6.5.4 Patch Analysis

In neither of the PCA or ISO patch statistics was a difference between burned and unburned treatment areas observed in the four image dates. Immediately following the fire the number of patches decreased in both burned and unburned treatments. This is may be a result of a flush of new vegetation growth covering a larger, more homogeneous area during the growing season because of the increased precipitation in 1998. The ISO patch statistics show a decrease in number of patches in 2002, which may also be a result of increased precipitation in 2002. Alternatively, the trend in patch statistics may be due to vegetation phenological differences throughout the growing season (Weber 2001, Viedma *et al.* 1997). Patch statistics may require a more fine scale data source to detect distinct trends.

6.5.4 Synthesis of Multi-temporal Data Characterizations

Comparing the three data characterization techniques used on the Landsat images for the land cover change analysis over time, the ISO and PCA classification results

support the expected increased difference between burned and unburned areas immediately following the fire, then decreased over time (Figure 1.2). The paired comparisons t-test and the change in distribution of pixels within the spectral classes each year support this. In contrast, no observable difference between treatment areas was found in the patch statistics for either classification technique. Similar to the patch statistics, the NDVI results as a whole did not reflect the expected trend in differences.

Different image acquisition dates, precipitation differences, and land management practices probably influenced the classification results between image dates. The 1997 image was acquired in late September, 1998 was acquired in late May, and both 2001 and 2002 images were acquired in early July (Table 4.1). Phenological differences in the vegetation influence classification (Weber 2001, Viedma *et al.* 1997). Lower digital numbers in all bands during 1997 was due to the vegetation senescence at the end of the growing season. 2001 was a drought year, thus vegetation may have been more similar to the fall of 1997 than the summer of 2002. By comparing the difference between treatments within one image, any treatment effect should have been noticeable with changes in that treatment effect comparable between different images. However, the specific vegetation differences between burned and unburned areas may have been masked by the coarse spatial resolution of the Landsat satellite image data. The 30 ground resolution cell may be too large to detect distinct vegetation distribution.

In 1998 and 2002, total annual precipitation was considerably higher than the 15-year average (Figure 1.3, Appendix A). In contrast, total annual precipitation volumes in 1997 and 2001 were below average. In the drier years a greater number of patches were observed as the vegetation would grow in a more sparse or patchy distribution in

response to the reduced precipitation. Monthly precipitation also varied within each year (Appendix A). This too would have influenced the phenology of the vegetation affecting the spectral classes identified. The reduced precipitation might have allowed a greater population of ground squirrel habitation of the landscape increasing landscape patchiness through their grazing and near surface burrowing activities. In contrast, the ground squirrel (*Spermophilus richardsonii*) population may be reduced in years when high floods occur that drown the mammals, such as in 2002. Differences in vegetation observed in the NDVI data appeared to be a function of variation in precipitation moreso than treatment differences. Similarly, the patch statistics seemed to reflect vegetation changes in response to precipitation; the greater the precipitation early in the growing season, the more lush the vegetation and thus, the larger the patch size. In contrast, the ISO and PCA spectral classes appeared to detect some vegetation differences beyond the variation introduced by precipitation.

Grazing management within the study area is also annually variable and thus, may have influenced the vegetation present at each site. For example, spring grazing tended to increase the presence of forbs, especially *Geum triflorum* Pursh. at site 6, and decrease the grass, while later season grazing increased the amount of grass and *Artemisia frigida* Willd. present. Many landowners removed grazing pressure from the riparian zones during the growing season immediately following the fire. Their land management values influenced the vegetation cover because they reintroduced cattle at different intervals after the fire, creating variable vegetation community composition and structure. Cattle are selective grazers and will graze areas with preferred forage, forming patches (Willms *et al.* 1988). Once cattle were reintroduced to the landscape, their familiarity with the

landscape may have led them back to the pre-fire grazing of preferred forage areas, thus forming grazing patches again. Alternatively, the fire may have removed the ungrazed patches thereby reducing any effects of cattle selectivity and more evenly distributing the grazing pressure.

Encroaching woody plants may also cause increased heterogeneity across the study area by increasing the number of patches present, or in contrast, may increase homogeneity of a site by increasing the patch size of current woody communities. The increased number of patches would result in a greater mix of spectral reflectance composing the pixel and thus lead to increased spectral confusion. Increased patch size would result in a more unified spectral reflectance decreasing spectral variability. Variable grazing management in riparian zones may also increase heterogeneity by creating differentially grazed patches where part of the landscape was more heavily grazed than others. In areas where grazing was lighter a greater recovery of willows was observed, while more heavily grazed areas were composed of shorter dense shrubs, such as *Symphoriocarpos occidentalis* Hook., *Rosa* L., and *Eleagnus commutata* L.

The complex interactions between image data acquisition, precipitation and land use management introduces unknown variation to the image analysis. Overall the ISO and PCA classifications seemed adequate for monitoring land cover change resulting from fire over time, however, the variability introduced by a wildfire, the natural environment and human activity were also present and most likely influenced spectral patterns observed.

6.6 Conclusions

The ISO and PCA spectral classes showed an increased difference between burned and unburned areas immediately following the fire that decreased over time (Figure 1.2). Multi-temporal Landsat data can be used to detect general changes in riparian zone vegetation distribution over time (Hypothesis #4), but not species composition.

6.7 Literature Cited

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Chapter 7 Synthesis

7.1 Summary and Conclusions

The purpose of this study was to isolate riparian zones within Landsat satellite image data and assess the vegetation difference between burned and unburned areas over time. Specific objectives were 1) to assess the field vegetation composition in 2001 and 2002, 2) to register all digital data into one projection for an accurate multi-temporal study, 3) compare ISO, NDVI and PCA spectral classes to field classes, and 4) apply ISO, NDVI and PCA in a multi-temporal analysis of landscape change.

Field data were collected in the riparian zones to document percent non-woody vegetation, woody vegetation and other cover. Results showed no significant difference between treatments in 2001, and a significant increase in woody vegetation cover between treatments in 2002. From 2001 to 2002, percent woody vegetation cover increased. This increased cover of woody vegetation may be due to grazing management by the local ranchers. Continuous grazing throughout the season results in a greater grazing pressure on palatable vegetation such as grasses and forbs, reducing their percent cover. In turn, unpalatable vegetation such as shrubs mature becoming more unpalatable or spreading, reducing the extent of palatable vegetation (Anderson and Bailey 1979, Archibald *et al.* 1998). Overall, there were few vegetation differences between treatments (Hypothesis #1).

After completion of the field assessment, riparian zones were delineated using a combination of aerial photographs and the satellite image data. Digital data were registered to within half a pixel, which is necessary for an effective comparison of datasets (Lillesand and Kiefer 2000). The riparian zones were delineated with an overall

RMSE of less than one pixel and the images visually agreed when aligned (Hypothesis #2).

From the transect data, 10 detailed field classes based on percent cover of bare soil, percent cover of shrubs and vegetation height were identified. The ISO and PCA characterization techniques used were programmed to detect 10 spectral classes to correlate with the field classes. Neither the ISO nor the PCA spectral classes directly related to individual specific vegetation classes, but included multiple field classes suggesting they were not spectrally separable (Lillesand and Kiefer 2000). The NDVI contingency table showed relationships between vegetation classes and NDVI values, but the spectral patterns overlapped considerably, preventing formation of NDVI classes. None of the characterization techniques used specifically represented the vegetation cover in the field suggesting that a different characterization technique is necessary to identify field observed vegetation differences in Landsat satellite image data. The field data trends from 2001 to 2002 were identified in the ISO and PCA derived spectral class data, but not NDVI characterization (Hypothesis #3).

The multi-temporal analysis using the three characterization techniques involved an analysis of all selected riparian zones in the four image dates. Similar trends were observed in both the ISO and PCA classifications techniques with respect to differences between burned and unburned areas over time (Figure 7.1). However, the NDVI analysis showed no strong trends or differences between burned and unburned areas (Figure 7.1). The ISO and PCA classifications of multi-temporal Landsat satellite image data may be used to monitor changes in vegetation distribution over time (Hypothesis #4).

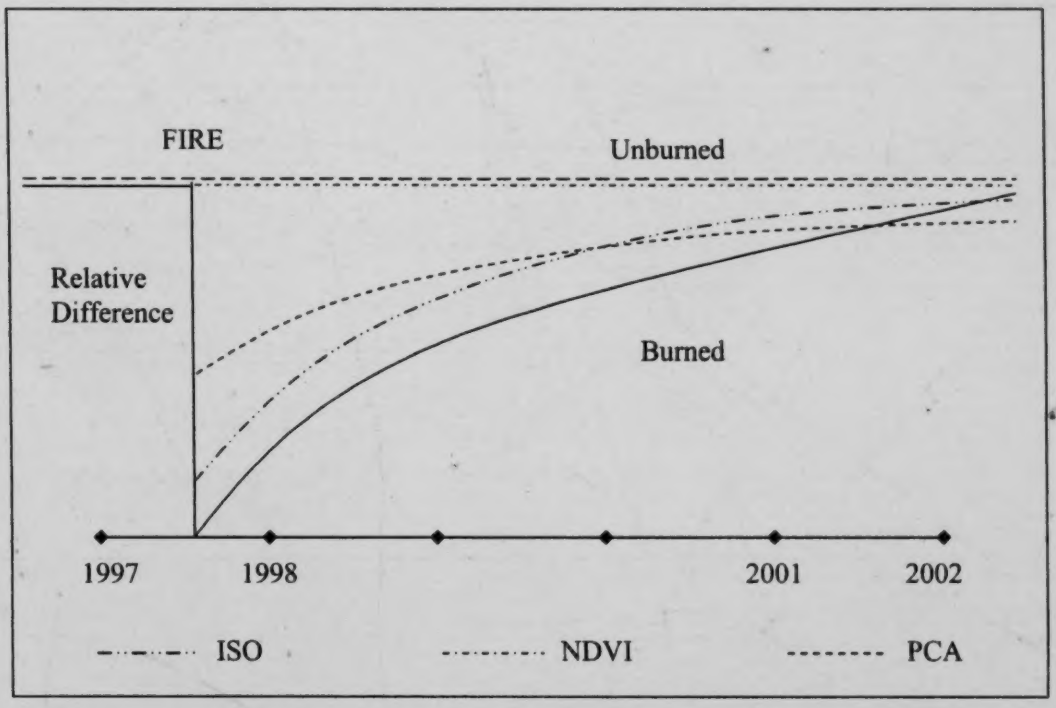

Figure 7.1 An illustration of mean vegetation differences between burned and unburned areas following fire based on ISO, NDVI and PCA data characterization techniques.

Overall, this study demonstrated the limitations of multi-temporal Landsat satellite image data as a tool to monitor changes in vegetation differences between selected burned and unburned riparian zones in the foothills rough fescue grasslands of southwestern Alberta. The ISO and PCA classification maps were very informative about changes in vegetation distribution over time. However, it is unknown whether the changes in vegetation distribution were due solely to fire effects. It is probable that the changes in vegetation distribution were also the result of the complex interaction between the fire, post-fire grazing management and post-fire weather conditions. This fire was a natural, unplanned disturbance, and thus was very difficult to study because of the lack of

benchmark data (Turner *et al.* 1997, Pyne *et al.* 1996). Grazing management for this landscape included continuous and rotation cattle grazing, but cattle grazing distribution across each pasture is variable. Pre-fire grazing patterns by cattle may have been different than post-fire, however, neither of these were recorded consistently. When grazing occurs on post-fire landscapes the ability to detect change is decreased (Oleson *et al.* 1995). The fire severity across the riparian zones was highly variable (Wowk 2000), and thus post-fire conditions such as site moisture, soil albedo and soil temperature would be diverse across the landscape.

The precipitation record for 1998 and 2002 indicated an above-average amount of precipitation, while 1999 to 2001 were considered drought years by local ranchers. Thus, the greater precipitation in 1998 facilitated vegetation recovery (Bork *et al.* 2000), but the subsequent drought may have increased variability of vegetation recovery across riparian zones. The satellite images available for the study were not anniversary dates due to limitations such as overall image quality and cloud cover. Some authors (Weber 2001, Viedma *et al.* 1997) suggest that similar vegetation phenology is more important than anniversary dates, however the complications involved in acquiring images for specific vegetation phenology during a growing season are similar to acquiring anniversary dates. Alternatively, the ground resolution of Landsat satellite image data is 30 m, which may be too coarse for monitoring specific changes in vegetation in riparian zones. Studying rangelands with remotely sensed data is complicated and requires consideration of many variables including weather, grazing and natural disturbances, as well as their interactions.

7.2 Suggestions for Further Research

Multi-temporal Landsat satellite image data were applied to detect and monitor vegetation change in selected riparian zones across foothill landscapes. However, further refinement of analysis technique is necessary to match spectral classes to ecologically important field classes.

- Satellite image data for 1999 and 2000 would have been good to fill in some blanks and indicate whether vegetation distribution between burned and unburned treatments were similar to pre-fire condition within 1 and 2 years of the fire, even though no field data were available for those years.
- Given that riparian zones are very narrow, patchy features within the rangeland, perhaps sensors with finer spatial resolution than Landsat may aid in identifying vegetation classes. However, increased spatial resolution will increase the cost of data acquisition and the opportunity for a multi-temporal dataset may not be possible. Furthermore, all remote sensing systems have problems with mixed pixels (Wylie *et al.* 1996).
- Techniques for spectral unmixing (Garcia-Haro *et al.* 1999 and Casals-Carrasco *et al.* 2000) have been developed and are more widely used, which may increase vegetation identification using Landsat satellite image data. However, identification of pure end-members may be difficult in these heterogeneous riparian zones
- Including a digital elevation model in the classification technique may have increased spectral pattern correlation with field classes by further separating vegetation according to topography, which played an important role due to the sharp environmental gradients present in riparian zones.

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Appendix A Precipitation Data

Table A.1 Precipitation (mm) at the Five Mile Station, Alberta (Figure 1.1), where the 1986-2001 15-year annual average is 511.6 mm (Nadeau 2002, Personal Communication).

Month	Year					
	1997	1998	1999	2000	2001	2002
January	18.9	29.1	9.1	24.7	8.5	29.4
February	19.9	2.0	0.6	16.0	13.9	62.7
March	40.6	76.1	27.7	31.2	10.7	73.6
April	52.9	42.2	56.5	27.3	119.3	38.0
May	144.7	187.9	56.0	13.2	0.0	114.5
June	56.3	225.7	93.6	60.2	106.9	269.8
July	29.0	58.9	24.3	16.1	31.4	50.5
August	55.7	25.9	36.6	31.6	4.3	114.3
September	24.5	35.8	32.1	70.4	12.8	83.6
October	15.9	6.2	27.6	7.0	8.0	26.3
November	5.0	17.4	22.5	5.6	17.7	3.5
December	5.4	31.9	12.6	19.9	6.7	30.6
Total Annual	468.8	739.1	399.2	323.2	340.2	898.8

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Nadeau, D. 2002. Personal Communication. Precipitation data for Five Mile Creek Station. Fort McLeod, Alberta, Canada. (January 30, 2003)

Appendix B Field Cover Frequency Histograms

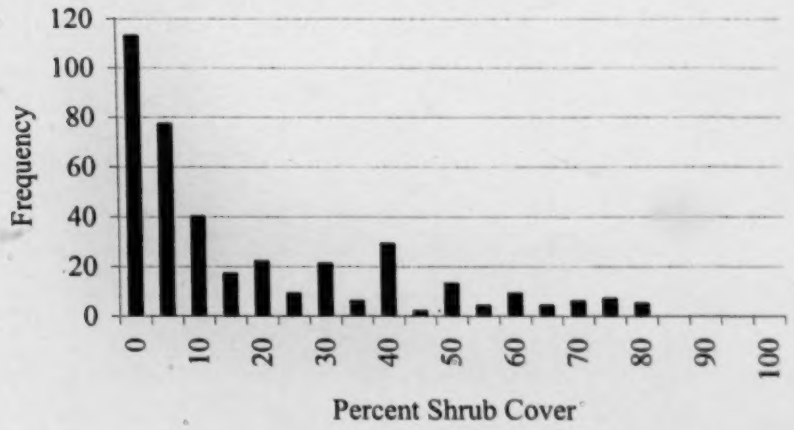

Figure B.1 Frequency of percent shrub cover across plant communities for 2001 and 2002.

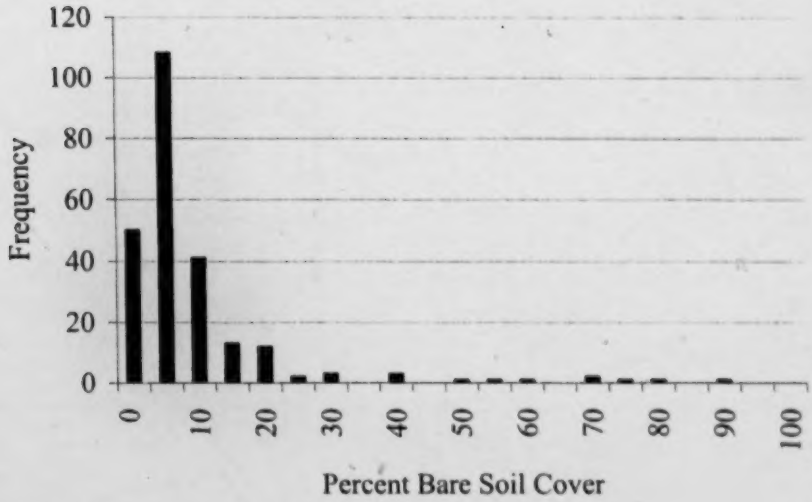

Figure B.1 Frequency of percent bare soil cover across plant communities for 2001 and 2002.

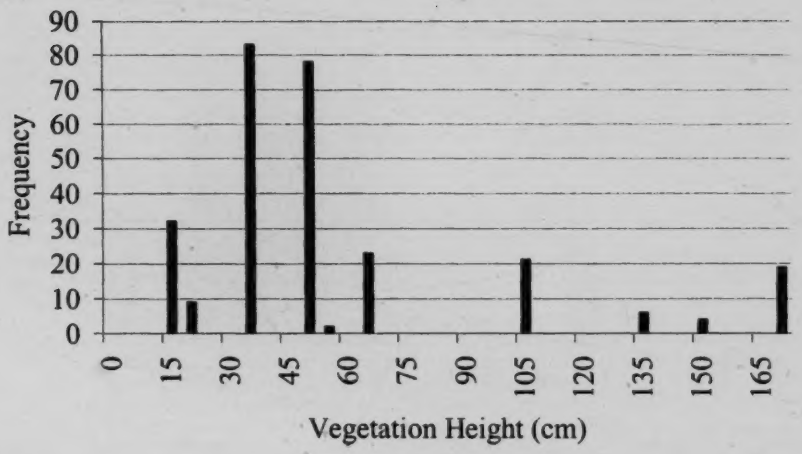

Figure B.1 Frequency of average vegetation height in plant communities for 2001 and 2002.

Appendix C Covariance Values for the Isocluster Classifications

Table C:1 Covariance of 1997 ISO derived clusters.

Class 1			**Class 5**		
	Band 3	Band 4		Band 3	Band 4
Band 3	6.03656	1.93441	Band 3	2.64653	-0.7951
Band 4	1.93441	5.18925	Band 4	-0.7951	2.6351
Class 2			**Class 6**		
	Band 3	Band 4		Band 3	Band 4
Band 3	4.8033	0.4542	Band 3	11.6439	-1.2051
Band 4	0.4542	3.8003	Band 4	-1.20513	8.53846
Class 3			**Class 7**		
	Band 3	Band 4		Band 3	Band 4
Band 3	2.93245	-1.0244	Band 3	5.45884	0.05466
Band 4	-1.0244	2.56811	Band 4	0.05466	6.21862
Class 4					
	Band 3	Band 4			
Band 3	3.82796	-0.6011			
Band 4	-0.6011	4.67312			

Table C.2 Covariance of 1998 ISO derived clusters.

Class 1			**Class 5**		
	Band 3	Band 4		Band 3	Band 4
Band 3	11.44762	0.78810	Band 3	8.25075	-0.52327
Band 4	0.78810	17.29048	Band 4	-0.52327	4.36036
Class 2			**Class 6**		
	Band 3	Band 4		Band 3	Band 4
Band 3	8.84038	1.47500	Band 3	14.77239	2.84815
Band 4	1.47500	6.30192	Band 4	2.84815	6.94074
Class 3			**Class 7**		
	Band 3	Band 4		Band 3	Band 4
Band 3	8.87121	1.83239	Band 3	31.76087	9.72826
Band 4	1.83239	6.71591	Band 4	9.72826	25.69384
Class 4			**Class 8**		
	Band 3	Band 4		Band 3	Band 4
Band 3	12.69540	-1.40805	Band 3	34.34280	4.59186
Band 4	-1.40805	15.82644	Band 4	4.59186	42.99621

Table C.3 Covariance of 2001 ISO derived clusters.

Class 1			**Class 5**		
	Band 3	Band 4		Band 3	Band 4
Band 3	10.7536	1.46377	Band 3	12.2143	0.04355
Band 4	1.46377	14.4765	Band 4	0.04355	9.5151
Class 2			**Class 6**		
	Band 3	Band 4		Band 3	Band 4
Band 3	9.84366	0.07817	Band 3	8.41844	-0.41844
Band 4	0.07817	12.4283	Band 4	-0.41844	5.07801
Class 3			**Class 7**		
	Band 3	Band 4		Band 3	Band 4
Band 3	9.54	1.42	Band 3	28.2253	5.70751
Band 4	1.42	6.72154	Band 4	5.70751	17.2569
Class 4			**Class 8**		
	Band 3	Band 4		Band 3	Band 4
Band 3	7.36111	1.95397	Band 3	27.37	-4.82258
Band 4	1.95397	7.53968	Band 4	-4.82258	13.5484

Table C.4 Covariance of 2002 ISO derived clusters.

Class 1			**Class 4**		
	Band 3	Band 4		Band 3	Band 4
Band 3	10.5116	-0.2805	Band 3	10.5836	0.60164
Band 4	-0.28049	22.4024	Band 4	0.60164	19.4098
Class 2			**Class 5**		
	Band 3	Band 4		Band 3	Band 4
Band 3	14.9319	1.75452	Band 3	26.2384	-3.67877
Band 4	1.75452	10.0506	Band 4	-3.67877	38.6231
Class 3			**Class 6**		
	Band 3	Band 4		Band 3	Band 4
Band 3	14.2944	-3.6127	Band 3	8.32698	1.04125
Band 4	-3.6127	9.87302	Band 4	1.04125	11.9663